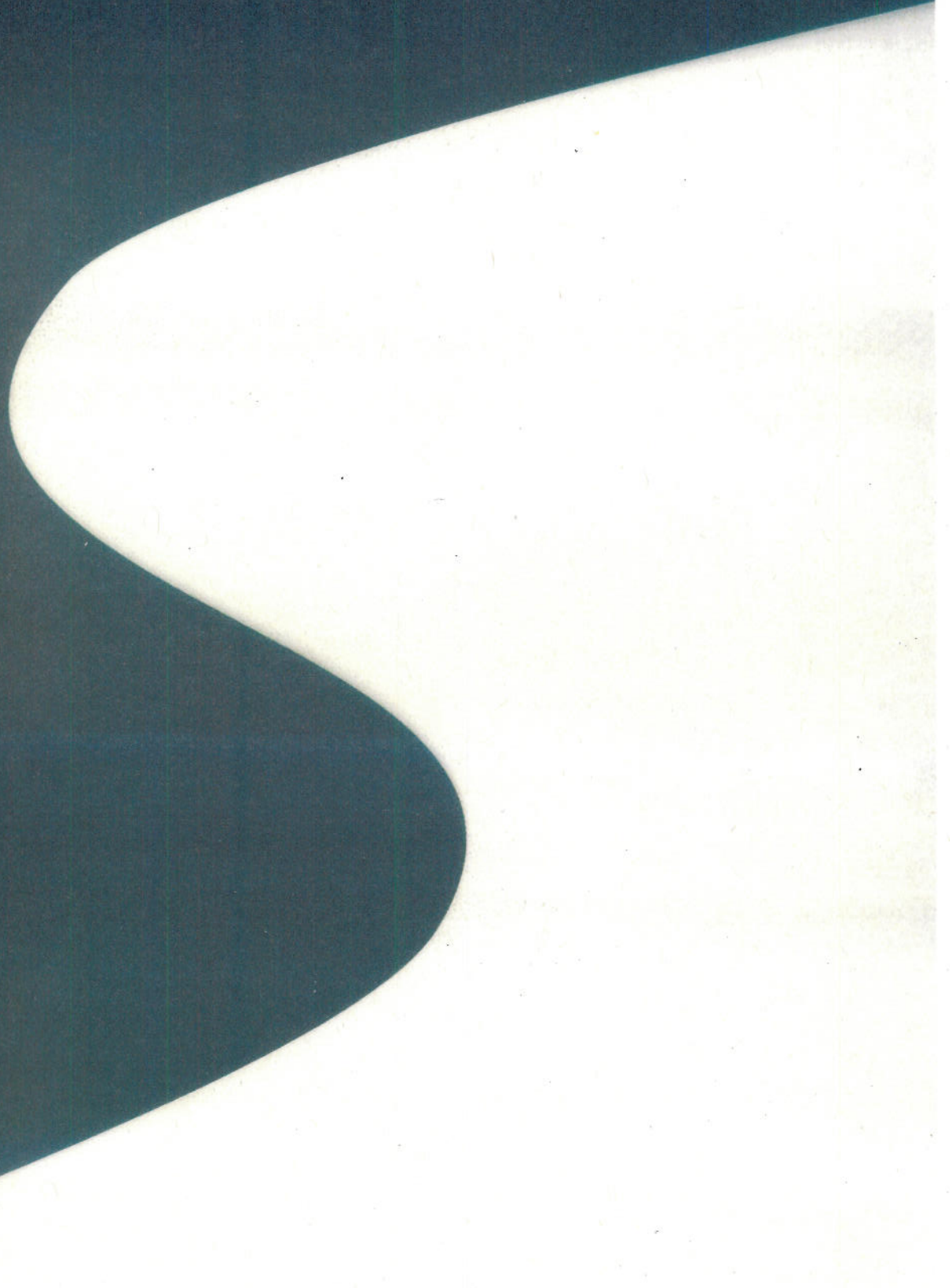


Stratus Consulting



**Habitat-Based Replacement Costs:
An Ecological Valuation of the Benefits
of Minimizing Impingement and
Entrainment at the Cooling Water
Intake Structure of the Pilgrim Nuclear
Power Generating Station in Plymouth,
Massachusetts**

Prepared for:

The New England Interstate
Water Pollution Control
Commission and
The U.S. Environmental
Protection Agency, Region 1

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February 5, 2002

SC10026

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Acronyms

BADCT	best available demonstrated control technology
BAT	best available technology
BMPs	best management practices
BPT	best practicable technology
BTA	best technology available
CSO	combined sewer overflow
CWIS	cooling water intake structure
EAM	Equivalent Adult Model
EPA	U.S. Environmental Protection Agency
HEA	habitat equivalency analysis
HRC	habitat-based replacement cost
HSI	Habitat Suitability Indices
I&E	impingement and entrainment
NERR	Naragansett Estuarine Research Reserve
NPDES	National Pollutant Discharge Elimination System
NPS	nonpoint source
SAV	submerged aquatic vegetation

1. Introduction and Regulatory Context

1.1 Introduction

Cooling water intake structures (CWISs) are regulated by the U.S. Environmental Protection Agency (EPA or Agency) and the States, pursuant to Section 316(b) of the Federal Water Pollution Control Act (Clean Water Act) [33 U.S.C. § 1326]. Section 316(b) requires that adverse environmental impacts such as impingement and entrainment (I&E) of aquatic organisms be minimized by requiring the best technology available (BTA) at CWISs.

The Agency is developing national standards under Section 316(b) for new and existing facilities. Furthermore, the Agency is reissuing the National Pollutant Discharge Elimination System (NPDES) permit for the Pilgrim Nuclear Power Generating Station in Plymouth, Massachusetts. Because both the costs of BTA and the benefits of minimizing adverse environmental impacts can be substantial, the Agency is developing site-specific information about the costs and benefits of BTA at CWISs. Therefore, the public, the Agency, and the regulated community have much at stake to ensure that complete and accurate cost and benefit information is incorporated into the national rulemaking and NPDES permits.

Unfortunately, complete information about the costs of BTA has been easier to obtain, usually with the help of the regulated community, than complete information about the benefits of minimizing I&E losses. Conventional techniques to value the benefits of technologies that reduce I&E losses at Section 316(b) facilities often omit important ecological and public services. In contrast, the habitat-based replacement cost (HRC) method can be used in benefit-cost analyses to value a broad range of ecological and human services affected by I&E losses that are either undervalued or ignored by conventional valuation approaches.

1.2 Regulatory Context

Congress enacted Section 316(b) of the Clean Water Act because of fish kills at power plant CWISs preceding the 1972 enactment. Fish kills are still the primary environmental impact of CWISs. Section 316(b) provides that any standard established pursuant to Sections 301 or 306 of the Clean Water Act and applicable to a point source must require that the location, design, construction, and capacity of CWISs reflect the BTA for minimizing adverse environmental impacts. Section 316(b) applies to the intake of cooling water rather than its discharge, which is regulated separately under Sections 301, 306, and 316(a) of the Clean Water Act. The two parts of Section 316 are related because the BTA used to address intake losses under Section 316(b) usually affects the thermal discharges regulated by Section 316(a).

Following settlement of a lawsuit, the Agency is developing national standards, pursuant to Section 316(b), in three phases: Phase I for new facilities, Phase II for existing electric generating plants that use large amounts of cooling water, and Phase III for electric generating plants using smaller amounts of cooling water and for manufacturers. In the mean time, the Agency and the States issue NPDES permits with BTA requirements for facilities with CWISs on a site-by-site basis. The Pilgrim facility is an example of a facility with CWISs covered by Section 316(b) for which the Agency will reissue the NPDES permit with BTA requirements (Massachusetts has not requested NPDES authority from the Agency).

BTA is a standard that specifies limits that are uniform, technology based, and technology forcing. Clean Water Act Sections 301, 304, 306, and 316(b) all require establishment of regulatory limitations based on uniform technology that minimize impacts locally. The Agency has promulgated best available technology (BAT), best practicable technology (BPT), and best available demonstrated control technology (BADCT) for the discharge of pollutants by the steam electric generating industry at 40 C.F.R. Part 423 (47 F.R. 52290).

1.3 Habitat-Based Replacement Costs

Conventional valuation techniques, such as those that focus on recreational and commercial fishing losses, omit important ecological and public services by relying on direct use values of impacted fish targeted by recreational and commercial anglers. However, many I&E losses are often eggs and larvae vital to the ecological system but with no obvious direct use values. Some Section 316(b) facilities may have relatively small numbers of species and life stages that are targeted by anglers, so commercial and recreational losses may be only a small subset of the species lost to I&E. Moreover, for the species that are targeted by recreational or commercial anglers, the reliance on adult equivalents omits the ecological services and associated public values provided by early life stages that do not make it to adulthood in the environment. Another conventional valuation technique bases the value of I&E losses on the costs of restoring aquatic organisms using hatchery and stocking programs. However, the cost of restoring fish through stocking does not address a number of ecological services, and addresses others inefficiently.

In contrast, the HRC valuation technique is based on the cost of offsetting I&E losses by increasing fish production through habitat creation, restoration, or enhancement. HRC can be used in benefit-cost analyses to value a broad range of ecological and human services associated with I&E losses that are either undervalued or ignored by conventional valuation approaches. Economists and policy makers have long recognized that the public places value on environmental benefits well beyond beneficial impacts on direct uses, but much of the professional literature focuses on recreational and other direct use values derived from the commercial and recreational impacts valuation method. In contrast, the HRC method defines the value of all I&E losses as the expenditures that would be required to replace all organisms lost to I&E at a CWIS through enhanced natural production in the environment. In short, the HRC method values lost resources by the costs of the programs required to naturally replace

those same resources. The replaced organisms would then be available not only for commercial and recreational human use but also as prey for a wide range of aquatic and terrestrial organisms, as well as the full range of complex ecological functions provided by those organisms. As a result, by focusing on replacement of natural habitats, the HRC method values fish and other organisms that are truly equivalent to those lost by allowing species to reproduce in their natural habitats using their native strategies (as opposed to most fish stocking programs). In addition, because the HRC results are based on the natural replacement of all relevant species, life stages, behaviors, and ecological interactions, for as long as the habitats remain viable, the resulting valuations of I&E losses effectively incorporate the complete range of ecological and human services, even when those services are difficult to measure or poorly understood.

1.4 Organization of this Report

Chapter 2 describes the Pilgrim facility, the facility's environmental setting in Cape Cod Bay near the mouth of Plymouth Bay, and the major environmental stressors near the facility and in the bay. Chapter 3 explains the need for new techniques to value more comprehensively the benefits of minimizing I&E, explains how the HRC method fills this need, presents how the HRC method works, and discusses the strengths and weaknesses of the HRC method. Chapter 4 describes each of the eight HRC steps as they were applied to the Pilgrim facility, and the results of the HRC analysis for this facility.

A companion report of the HRC method applied to the Brayton Point Station in Somerset, Massachusetts was also prepared for the New England Interstate Water Pollution Control Commission and the Agency.

2. Overview and Environmental Setting of the Intake Facility

2.1 Location and Description of the Pilgrim Facility

Pilgrim is a 670 MW nuclear power plant located in Plymouth, Massachusetts (Figure 2-1). Commercial operation of the Pilgrim station began in 1972 (ENSR, 2000). The mouth of Plymouth Bay is approximately 4 miles northwest of the Pilgrim site. Pilgrim uses water from the surrounding water bodies as a coolant, and as water is drawn into the facility, aquatic organisms are entrained into the plant or are impinged on screens across the intake pipes.

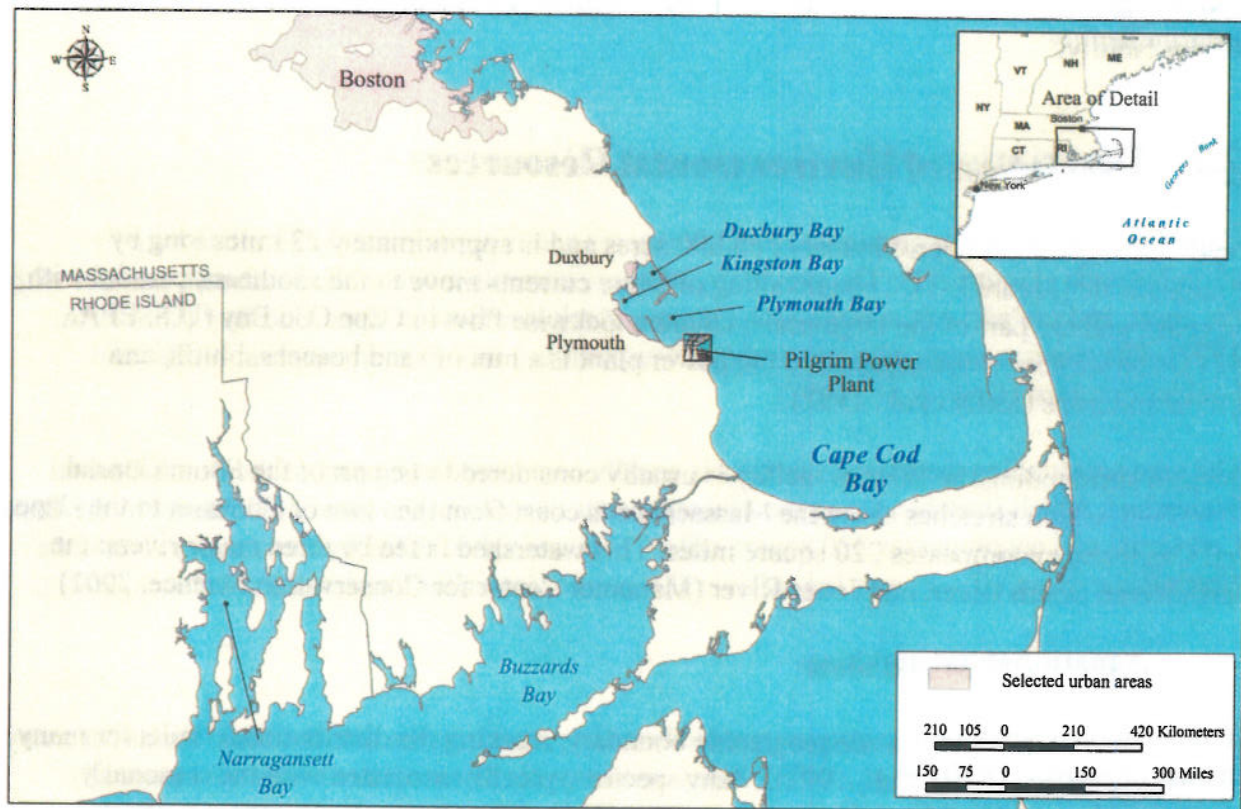


Figure 2-1. Location of the Pilgrim Nuclear Power Generating Station in Plymouth, Massachusetts.

The Pilgrim facility contains two water-moderated, boiling water nuclear reactors with once-through condenser cooling systems. Water used for cooling the condenser is withdrawn from Cape Cod Bay through an artificially created intake embayment that is bounded by breakwaters and rip-rap (Tetra Tech, 2001). The entrance to the intake structure is 24 ft below sea level, and consists of wing walls, a skimmer wall, vertical trash racks, and traveling screens (Tetra Tech, 2001). The skimmer walls and trash racks are designed to remove large debris. Fish-escape openings are located in the skimmer walls and at the end of each intake structure. Traveling screens are designed to remove some organisms and smaller debris, and they consist of wire mesh with 0.25 by 0.50 in. openings. Material caught on the traveling screens is backwashed first with low pressure water to remove organisms, followed by a high pressure wash to prevent heavy fouling (Tetra Tech, 2001).

A number of intake technology alternatives have been proposed at the Pilgrim facility, including behavioral barriers, diversion devices, alternate intake screen systems, and flow reduction technologies (Tetra Tech, 2001). None of these technologies have been selected for use at the Pilgrim facility.

2.2 Description of Environmental Resources

Cape Cod Bay covers approximately 365,000 acres and is approximately 23 miles long by 23 miles wide (Figure 2-1). The prevailing offshore currents move to the southeast, parallel with the coast, and are part of the large-scale, counterclockwise flow in Cape Cod Bay (U.S. EPA, 1977). The western shore adjacent to the power plant is a mix of sand beaches, bluffs, and boulder outcrops (Kelly et al., 1992).

The area surrounding the Pilgrim facility is usually considered to be part of the South Coastal Watershed, which stretches along the Massachusetts coast from the town of Cohasset to the Cape Cod Canal and encompasses 220 square miles. This watershed is fed by three major rivers: the North River, South River, and Jones River (Manomet Center for Conservation Science, 2001).

Aquatic habitat and biota

In this region, Cape Cod is a zoogeographic boundary, marking the distributional limits for many marine organisms (Kelly et al., 1992). Many species typically associated with the seasonally warmer waters south of Cape Cod, such as spotted hake (*Urophycis chus*), oyster toadfish (*Opsanus* spp.), and rainwater killifish (*Lucania parva*), occasionally move north into Cape Cod Bay in mid- to late summer. However, most northern species, such as rainbow smelt (*Osmerus mordax*), Atlantic tomcod (*Microgadus tomcod*), and rock gunnel (*Pholis gunnellis*), rarely extend into the waters south of Cape Cod (Able and Fahay, 1998). Commercially and recreationally important species found in the waters near the Pilgrim station include winter flounder (*Pseudopleuronectes americanus*), Atlantic menhaden (*Brevoortia tyrannus*), and Atlantic herring (*Clupea harengus*) (Kelly et al., 1992). Forage species, such as cunner

(*Tautoglabrus adspersus*) and Atlantic silverside (*Menidia menidia*), are also found in the waters near the Pilgrim station (Entergy, 2000).

The area surrounding the Pilgrim facility supports a wide variety of habitats, including open sandy and rocky bottoms, seagrass beds, salt marshes, tidal mud flats, sandy beaches and dunes, coastal ponds, and open water. Plymouth Bay supports a considerable amount of eelgrass (*Zostera marina*) habitat (Manomet Center for Conservation Sciences, 2001). Eelgrass provides an important source of food and refuge for a number of species in the area, including Atlantic cod (*Gadus morhua*), pollock (*Pollachius virens*), and threespine stickleback (*Gasterosteus aculeatus*).

The benthic community of Cape Cod Bay near Plymouth consists mainly of annelids; elsewhere it is diverse. Immediately adjacent to the Pilgrim facility, the red algae, Irish moss (*Chondrus crispus*), is abundant on the sea floor (Entergy, 2000). At the outfall of the Pilgrim facility's discharge canal, the Irish moss is noticeably denuded, or sparse and stunted, which may be a result of sensitivity to thermal effluents, chemical discharge of chlorine, or scouring by high velocity flows near the facility's cooling water discharge outfall (Entergy, 2000).

Marine shore-zone fishes such as Atlantic silverside, mummichog (*Fundulus heteroclitus*), striped killifish (*Fundulus majalis*), Atlantic herring, sand lance (*Ammodytes* spp.), blueback herring (*Alosa aestivalis*), alewife (*Alosa pseudoharengus*), and winter flounder occupy the intertidal and shallow subtidal zones near the Pilgrim station. Many of these shore-zone fishes are important as forage for piscivorous fishes, birds, and invertebrates. The close proximity of these species to shore makes them more susceptible to power plant intake and discharge activities.

Many anadromous species of fishes are found in the vicinity of the Pilgrim facility. These species include alewife, Atlantic herring, Atlantic tomcod, blueback herring, rainbow smelt, and white perch (*Morone americana*). Rivers that support anadromous fish spawning include the Eel River, Jones River, Bluefish River, and Green Harbor Creek (Massachusetts Audubon Society, 2001). Cape Cod Bay and the Gulf of Maine also support a variety of marine mammals, including whales, porpoises, and seals (Conkling, 1995).

Threatened, endangered, and other rare, declining, or vulnerable species

The area surrounding the Pilgrim facility supports several threatened or endangered species, as well as species of special concern that have suffered declines and could easily become threatened. Threatened and endangered species and species of special concern that occur in and around the town of Plymouth include birds such as the piping plover (*Charadrius melodus*), least bittern (*Ixobrychus exilis*), bald eagle (*Haliaeetus leucocephalus*), barn owl (*Tyto alba*), roseate tern (*Sterna dougallii*), least tern (*S. antillarum*), common tern (*S. hirundo*) and Arctic tern (*S. paradisaea*). Listed reptile species include the red-bellied turtle (*Pseudemys rubriventris*), spotted turtle (*Clemmys guttata*), and eastern box turtle (*Terrapene carolina*). In addition, the

tidewater mucket (*Leptodea ochracea*), the triangle floater (*Alasmidonta undulata*), and the bridle shiner (*Notropis bifrenatus*) are species of special concern in this area (NHESP, 2001).

Birds

The Plymouth Bay area has been listed as an "Important Bird Area" by the Massachusetts Audubon Society. This area supports a large colony of terns (including roseate, least, common, and Arctic terns), a large heronry on Clark's Island, and many species of migratory and wintering shorebirds and waterfowl (Massachusetts Audubon Society, 2001). Terns are often considered an indicator of marine ecosystem health. They eat small fish, including small herring, hake, sand eels, butterfish (*Peprilus triacanthus*), and young bluefish (*Pomatomus saltatrix*). When populations of small marine fishes are threatened, terns may also face starvation (Conkling, 1995).

Fisheries

Massachusetts has a long-standing tradition of recreational and commercial marine fishing. Popular recreational targets include bluefish, Atlantic cod, summer flounder (*Paralichthys dentatus*), scup (*Stenotomus chrysops*), striped bass (*Morone saxatilis*), and Atlantic mackerel (*Scomber scombrus*). An estimated 17 million fish were caught in Massachusetts in 2000 by recreational anglers. Commercial fisheries include Atlantic cod, winter flounder, yellowtail flounder (*Limanda ferruginea*), goosefish (*Lophius americanus*), haddock (*Melanogrammus aeglefinus*), Atlantic herring, and many others. Shellfishing for ocean quahog clams (*Mercenaria mercenaria*), deepsea red crab (*Paralomis granulosa*), American lobster (*Homarus americanus*), and bay scallops (*Argopecten irradians*) is also an important source of revenue in Massachusetts. In 2000, commercial fishing revenues in Massachusetts totaled more than \$120 million, and commercial shellfishing revenues totaled more than \$288 million (NMFS, 2001).

Tourism

A multitude of scenic and cultural resources in and along the Massachusetts bays attract tourists from around the world. Plymouth County, where the Pilgrim power station is located, is one of the leading counties in Massachusetts in terms of tourism revenue. Plymouth Bay has approximately 55 miles of shoreline, including 16 miles of barrier beaches (Massachusetts Audubon Society, 2001). Plymouth Beach and Duxbury Beach are popular tourist attractions. Tourists can also visit Plymouth Rock and the National Monument to the Forefathers (Manomet Center for Conservation Sciences, 2001).

2.3 Major Environmental Stressors

Habitat alteration

Tidal restrictions have had a major impact on the salt marshes, ponds, and creeks within the communities of Duxbury, Kingston, and Plymouth. The Massachusetts Wetlands Restoration Program has listed 33 sites, encompassing approximately 200 acres, in these communities where wetlands are affected by tidal restrictions (MAPC, 2001). Tidal restrictions impede the flow of salt water into marsh areas, which can alter the hydrology of the site and result in changes to the flora and fauna. On shorelines and beaches, off-road vehicles also pose a threat to the coastal ecosystems. Use of the beaches and sand dunes by off-road vehicles destabilizes the dunes and impacts piping plover and tern colonies (Manomet Center for Conservation Sciences, 2001).

Non-native and invasive species

There are concerns over the introduction of non-native species into the coastal habitats of Massachusetts through ballast water on ships. One such species that has recently colonized southern Massachusetts waters is *Hemigrapsus sanguineus*, a crab native to the western North Pacific. *Hemigrapsus sanguineus* affects the local ecology by competing for food and habitat space. It eats a variety of algae and animals, including juvenile clams, and it may also be a food source for larger animals (MIT, 2000). It appears to occupy habitats very similar to native crabs in the region.

The most common invasive species at this site is *Phragmites australis*, a tall reed grass that grows in fresh and brackish waters and along the edges of salt marshes. Although *Phragmites* is native to much of New England, it can become invasive under certain conditions, choking out other plants and reducing valuable wildlife habitat. *Phragmites* thrives near freshwater inputs and in waters containing high levels of nutrients. *Phragmites* often becomes dominant in marshes that no longer receive adequate tidal flow as a result of backfilling, road construction, or erosion (Figure 2-2). Other invasive plant species found near the Plymouth facility include bittersweet (*Celastrus orbiculatus*) and saltspray rose (*Rosa rugosa*) (Manomet Center for Conservation Sciences, 2001).

Overfishing

Based on trends in catch and fishing effort, the U.S. Department of Commerce has stated that the dominant factor affecting commercial fish stocks is fishing. National Marine Fisheries Service statistics show that standardized trawl effort for groundfish in the Gulf of Maine has approximately doubled from 1976 to 1988. Despite the increasing efforts, fishermen have seen a decline in landings and catch per unit effort during the same period. The changes in commercial fish stocks brought about by overexploitation also have consequences for the noncommercial and recreational fish species prey species (Townsend and Larsen, 1992).



Figure 2-2. A stand of *Phragmites australis* in a tide-restricted salt marsh influenced by freshwater.

Source: MAPC, 2001.

Pollution

In 1988, 75% of Massachusetts' population resided in coastal counties (Gottholm and Turgeon, 1992). The high population density has made nonpoint source (NPS) pollution a major problem in the Massachusetts coastal area. When rainwater and snowmelt run over farm fields, city streets, lawns, and other surfaces, contaminants such as soil sediments, fertilizers, sewage, and pesticides are picked up and ultimately deposited into surface water. In many places, contaminated rainwater runs directly into coastal waters such as salt marshes and estuaries, impairing water quality and reducing the productivity of coastal habitats. Because estuaries serve as important breeding, nursery, and forage grounds for fish and other wildlife, commercial fisheries are ultimately affected by NPS pollution (CZM, 1994).

Excess loadings of nutrients is a particularly important pollution problem along the Massachusetts coast. These nutrient loadings are the most widespread factor altering the structure and function of aquatic systems by increasing macroalgal biomass and growth. Waquoit Bay National Estuarine Research Reserve on Cape Cod has experienced a particular problem with increases in seaweeds, which have reduced the extent of former eelgrass habitats (EHP, 2001).

3. Habitat-Based Replacement Cost Method

3.1 The Need for an Alternative to Conventional I&E Valuation Techniques

Conventional techniques to value the benefits of technologies that reduce I&E losses at Section 316(b) facilities can omit important ecological and public services. For example, valuations based on expected recreational and commercial fishing impacts rely on indirectly derived nonmarket value estimates (e.g., consumer surplus per angling outing as estimated by travel cost models) and direct market values, respectively. In both instances, all benefits are based solely on direct use values of the impacted fish, and the physical impacts are characterized by the adult life stage of the species targeted by the recreational and commercial anglers. However, at many Section 316(b) facilities, a large percentage of I&E losses are eggs and larvae, which are vital to a well functioning ecological system but have no obvious direct use values in and of themselves. Moreover, these facilities may have relatively small numbers of species and individuals that are targeted by anglers, so commercial and recreational losses may be only a small subset of the species lost to I&E. Even when losses of early life stages are included by conversion to adult equivalents, the ecological services and associated public values provided by early life stages that do not make it to adulthood in the environment are omitted.

Another conventional valuation technique bases the value of I&E impacts on the costs of restoring aquatic organisms using hatchery and stocking programs. However, the cost of restoring fish through stocking does not address a number of ecological services, and addresses others inefficiently. Shortcomings associated with the use of hatchery and stocking costs to estimate the value of I&E losses include the following:

- ▶ Reliable stocking costs are available only for the few species targeted by existing hatcheries, and these tend to be the same species addressed by recreational and commercial fishing valuations.
- ▶ The reported costs often do not include transportation costs.
- ▶ The costs associated with hatchery and stocking programs do not include the value of many ecological services affected by I&E losses, because hatchery fish are released at different life stages, in different numbers, and in different places than they would be produced in the natural environment.
- ▶ Hatcheries usually produce naive fish, which do not function as well as wild fish in the environment.
- ▶ Hatchery fish lack genetic diversity and disease resistance compared to fish produced in the natural environment (Hilborn, 1992; Meffe, 1992).

- ▶ Hatchery and stocking programs must continue as long as I&E losses occur, whereas natural habitat produces fish indefinitely, once properly restored and protected.
- ▶ At a number of locations where fish stocking programs are in place, significant questions remain as to whether the programs actually supplement the native fish populations, and if they do, the extent to which this occurs.

3.2 HRC Coverage of a Broader Range of Services and Values

The HRC method can be used in benefit-cost analyses to value a broad range of ecological and human services associated with I&E losses that are either undervalued or ignored by conventional valuation approaches. Economists and policy makers widely acknowledge that the public values environmental benefits well beyond beneficial impacts on direct uses (e.g., Fisher and Raucher, 1984). While much of the professional literature, especially empirical investigations, focuses on recreational and other direct use values, most Americans value water resource protection and enhancement, including reduction of I&E losses, for reasons that go well beyond their desire for recreational anglers to enjoy a larger consumer surplus (or commercial anglers to enjoy greater producer surplus).

For direct use benefits such as recreational angling, the predicted change in the stock of a recreational fishery affects recreational participation levels and/or the value of an angling day. However, I&E losses affect the aquatic ecosystem and public use and enjoyment in many ways not addressed by typical recreational valuation methods, creating a gap between known disruption of ecological services and what economists usually translate into monetary values or anthropocentric motives. Examples of ecological and public services (Peterson and Lubchenco, 1997; Postel and Carpenter, 1997; Holmlund and Hammer, 1999) disrupted by I&E, but not fully addressed by conventional valuation methods, include:

- ▶ disruption of ecological niches and ecological strategies used by aquatic species
- ▶ disruption of organic carbon transfer through the food web
- ▶ disruption of energy transfer through the food web
- ▶ decreased numbers of ecological keystone, rare, or sensitive species
- ▶ decreased numbers of popular species that are not fished, perhaps because the fishery is closed
- ▶ decreased numbers of special status (e.g., threatened or endangered) species
- ▶ increased numbers of exotic or disruptive species that compete well in the absence of species lost to I&E

- ▶ decreased local biodiversity
- ▶ disruption of predator-prey relationships
- ▶ disruption of age class structures of species
- ▶ disruption of public uses other than fishing, such as diving, boating, and birding
- ▶ disruption of public satisfaction with a healthy ecosystem.

The HRC method differs fundamentally from the commercial and recreational impacts valuation method because the latter accounts for only those species and life stages that can be valued directly, such as those species targeted by recreational or commercial anglers. In contrast, the HRC method defines the value of all I&E losses as the expenditures that would be required to replace all organisms lost to I&E at a CWIS through enhanced natural production in the environment. In short, the HRC method values lost resources by the costs of the programs required to naturally replace those same resources. The replaced organisms would then be available not only for commercial and recreational human use but also as prey for a wide range of aquatic and terrestrial organisms, as well as the full range of complex ecological functions provided by those organisms. As a result, by focusing on replacement of natural habitats, the HRC method values fish and other organisms that are truly equivalent to those lost by allowing species to reproduce in their natural habitats using their native strategies. In addition, because the HRC results are based on the natural replacement of all relevant species, life stages, behaviors, and ecological interactions, for as long as the habitats remain viable, the resulting valuations of I&E losses effectively incorporate the complete range of ecological and human services, even when those services are difficult to measure or poorly understood.

3.3 How the HRC Works

The HRC method values natural resource losses based on the costs of ecological habitat-based restoration activities which are scaled to increase natural production as an offset to the I&E losses. Thus, HRC uses resource replacement costs as a proxy for the value of resources lost to I&E. The HRC method is thus a supply-side approach for valuing I&E losses in contrast to the more typically used demand-side valuation approaches (e.g., commercial and recreational fishing impacts valuations).

In addition to valuing a wider range of losses, the HRC method also provides regulators with information to evaluate any environmental restoration proposed by the permittee to voluntarily offset future I&E losses associated with a technology that may be permitted. This information comprises a prioritized set of restoration alternatives for each species affected by I&E, estimates of the potential benefits of implementing those alternatives, and estimates of the effective unit

costs for those alternatives. The steps required to implement an HRC valuation of I&E losses are presented in Figure 3-1.

While the HRC method is a new approach for valuing losses of aquatic organisms from a CWIS, it is consistent with and related to lost resource valuation techniques such as habitat equivalency analysis (HEA) that have been recognized by federal courts as appropriate for use in valuing lost resources (for examples, see U.S. District Court, 1997, and U.S. District Court, 1999). Further, the principle of offsetting resource and ecosystem losses through restoration actions is incorporated in other components of the Clean Water Act, such as those addressing the losses of wetland areas (i.e., Section 404). The following subsections discuss the steps for conducting an HRC valuation of I&E losses.

3.4 Steps in the HRC Valuation

3.4.1 Quantify I&E losses by species

The first step in an HRC valuation quantifies the I&E losses from a Section 316(b) facility. This defines a CWIS's impacts, including temporal variations when multiple years of data are available, and thereby defines the gains of aquatic organisms that restoration actions should achieve. However, the I&E analyses performed by EPA are limited by the I&E monitoring data available for each facility, and therefore do not include losses of species not targeted by monitoring programs. In addition, many species are often combined and reported as a genus, family, or group of families (e.g., flounder species) because of insufficient identification capability within the monitoring program. This generally means that the analysis underestimates the value of impinged and entrained species that were not the focus of the facility's monitoring. HRC partially alleviates this problem because restoration of habitats for species monitored is likely to benefit other species lost but not monitored.

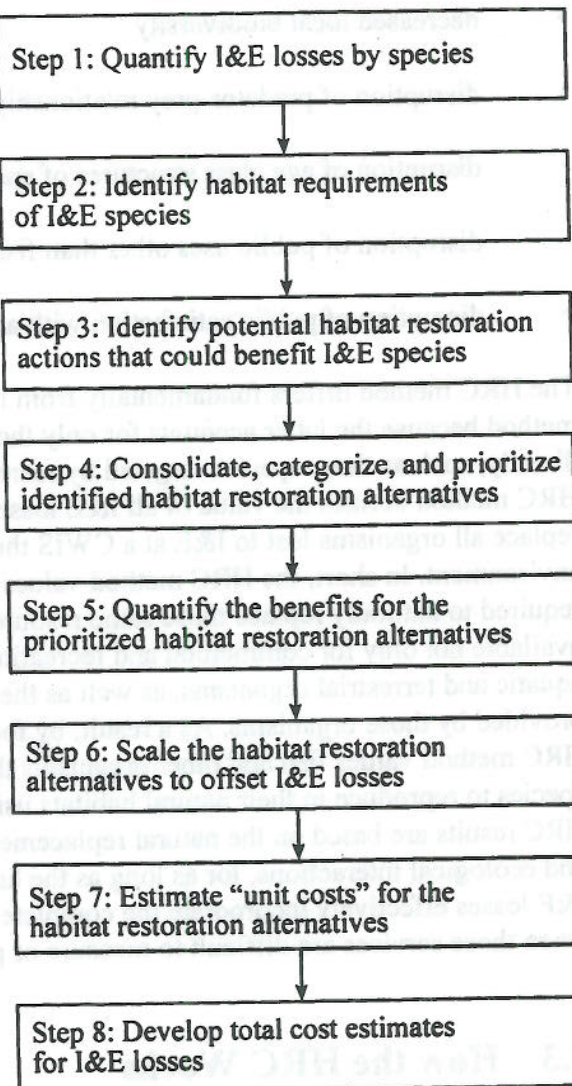


Figure 3-1. The 8 steps of the HRC method.

Because measured I&E losses often include multiple life stages (e.g., eggs, larvae, juveniles, adults) of any given species, total losses for each species are generally expressed as equivalent losses in a single, common life stage. This conversion is accomplished through the use of survival and production rates between life stages (younger life stages are always more abundant than older life stages because of mortality rates). A common life stage is generally chosen to facilitate the scaling of the restoration alternatives. For instance, early life stages are highly relevant for determining how much spawning habitat is required in cases where the productivity of spawning habitats is estimated. Adjusting the raw I&E loss data to a common life stage does not bias HRC results because many eggs are equivalent to fewer adults on both the I&E loss and the restoration gain side of the HRC equation. In other words, losing an adult to I&E is equivalent to losing many eggs because the adult represents survival through many life stages, but restoring an adult is equivalent to restoring many eggs for the same reason. Therefore, the life stage selected for reporting the losses should be highly relevant to the life stages affected by (and measurable in) restoration activities.

3.4.2 Identify habitat requirements of I&E species

The second HRC step identifies the habitat requirements of the aquatic organisms that are lost to I&E. A species' habitat requirements are usually identified through literature searches and discussions with local resource managers, biologists, conservationists, and restoration experts with specific knowledge of the species.¹ Local species characteristics and local habitat requirements and opportunities are used because of both biological variability and variation of local habitat conditions and constraints.

Because many I&E losses of aquatic organisms are realized in their earlier life stages (e.g., eggs, larvae, and juveniles), this step emphasizes habitat requirements for these early life stages, including spawning habitats. This emphasis is important because reducing constraints on adequate spawning is critical to increasing species production, is practical to achieve, and addresses directly the life stages that are most affected by impingement and entrainment.

3.4.3 Identify potentially beneficial habitat restoration alternatives

The third step in an HRC valuation identifies the habitat restoration alternatives that may increase the local production of the I&E species. As with identifying habitat requirements, this information is typically best developed through literature searches and discussions with local resource managers. In developing this information, special attention is paid to any remedial

1. For some species, very little may be known about life stage characteristics and habitat needs. In these cases, information about taxonomically related species or functionally related life stages may be used. Where relevant information is extremely limited, best professional judgment must be applied, including the possibility of omitting the species from the analysis.

action plans for local water bodies or local species management plans that present a series of projects or actions needed to address both specific and general constraints on the populations of aquatic organisms experiencing I&E losses.

This step must not be limited to restoration actions that have already been completed or that are already planned. While information about projects planned or under way is valuable, more comprehensive information about what restoration activities could improve the production of the affected species sufficient to fully offset I&E losses is essential to understand the full cost to society of I&E losses to the environment and the public. In other words, costs should only be constrained by biological understanding and engineering capability rather than existing funding and administrative opportunities.

While the difference between what is being done or planned and what could be done may in some cases be small, in other cases it may be quite significant. For example, in a location zoned for urbanized development, there may be little administrative opportunity for local wetland restoration. However, if available information and expert opinion suggest that increasing wetland acreage would be highly effective for increasing local production for a subset of affected species, a wetland restoration program should not be eliminated from consideration, even if such a program could not be implemented locally because of regulatory or administrative hurdles.

3.4.4 Consolidate, categorize, and prioritize identified habitat restoration alternatives

The fourth step in an HRC valuation consolidates the identified restoration alternatives, prioritizes them, and selects a preferred restoration alternative for each species.

The goal of consolidation is to eliminate redundancy in the proposals while producing a clearly defined set of restoration alternative categories for prioritization. In this step, specific project proposals, such as, "restore the 10-acre tract of former wetlands adjacent to marina X," are consolidated into more general categories for evaluation, such as "restore tidally connected *Spartina* marshes on the Massachusetts coast." This consolidation produces a more manageable set of restoration alternatives that can be evaluated against each other and costed.

The second part of this step, prioritizing the restoration alternatives, requires identifying a preferred alternative for each I&E species. This prioritization benefits from close coordination with local resource managers, both to define the criteria to rank the alternatives and to evaluate the alternatives against the criteria. One effective strategy for completing this task convenes relevant resource managers and stakeholders for an open review and discussion of the categorized restoration alternatives, with a goal of consensus on the preferred restoration alternative for each species with I&E losses.

3.4.5 Quantify the expected increases in species production for the prioritized habitat restoration alternatives

Quantifying the benefits of the preferred restoration alternatives to I&E species, the fifth HRC step, is critical for scaling the amount of restoration needed to offset calculated I&E losses. The best sources of data to quantify the benefits of a restoration alternative are rigorous, peer-reviewed studies that quantify the increases in production of I&E species that result from particular restoration activities. However, such studies are typically not available for many of the species that a particular facility impinges or entrains.

More commonly, the benefits of habitat restoration projects have to be estimated from species population densities measured or estimated in different habitats. The results of these studies are used to estimate increases in species production per unit of restored habitat by assuming that restoration provides similar habitat with similar productivity to that sampled. Estimates of the increased species production following restoration activities should account for lower initial (and perhaps permanent) productivity in restored versus pristine or unimpaired habitats (for a discussion of some of the factors that can affect productivity estimates in restored habitats, see Strange et al., in press). Again, local resource managers are essential to making realistic adjustments. In practice, these adjustments are usually integrated as a percentage of estimated baseline benefits in the HRC equation.

For some I&E species, neither restoration productivity data nor population density data by habitat are available. For these species, estimates of the increase in species production may be based on models of habitat-species relationships, such as Habitat Suitability Indices (HSI), data or studies on other habitats or other species with similar functional characteristics, or the best professional judgment of local resource managers.

3.4.6 Scale the habitat restoration alternatives to offset I&E losses

The sixth step scales the selected habitat restoration actions such that the magnitude of their expected increases in species production offsets the I&E losses. This step combines the estimated increases in species production associated with the restoration actions (step 5) with the quantified I&E losses (step 1). The scale of the required restoration (e.g., number of acres or feet of shoreline) is determined by dividing the I&E loss by the increase in species production produced by a unit area of habitat restoration. For example, if a facility's CWIS impinges and entrains 1 million year-one winter flounder per year, and local wetland restorations have been documented to produce 500 year-one winter flounder per acre per year (and wetland restorations are recognized as the most effective and cost-effective restoration alternative for winter flounder), then successful, sustained restoration of 2,000 acres of wetlands are required to offset these I&E losses.

The typical case involves I&E losses of multiple species, some of which have common preferred restoration alternatives and some of which do not. Where multiple species have the same preferred restoration alternative (e.g., restoration of tidal *Spartina* marshes), the appropriate scale to use in the HRC analysis is assumed to be the largest one among those species. In other words, if three species all benefit from the same restoration alternative and require 100, 500, and 1000 acres of *Spartina* habitat restoration to offset I&E losses, then 1000 acres is the value carried forward to the costing analysis. Although scaling the restoration alternatives in this way means that some species may be over-compensated, this approach is used because of the overriding principle that each species provides unique services and values, and losses of one species cannot be offset by gains in another. On the other hand, adjustments can be made to the required scale if the analysis is driven by a species whose I&E losses and/or restoration benefits are particularly uncertain or biased (e.g., the second-largest scale could then be selected). Such adjustments are made on a case-by-case basis and involve the prudent use of best professional judgement.

However, where multiple restoration activities are required to address all of the species, "collateral" benefits provided to a species by habitat restoration for a different species are included in the HRC analysis. Thus, the required scale of a preferred restoration alternative for a species may be reduced if it is benefitted by other kinds of restoration that are included to benefit other species. The amount of reduction necessary is estimated from the estimated collateral benefits provided by the other kinds of restoration.

3.4.7 Develop unit cost estimates

In the seventh step, the unit costs (e.g., costs per acre) for all preferred restoration alternatives are estimated. Unit cost estimates include all expenses associated with the design, implementation, administration, maintenance, and monitoring of each restoration alternative. These costs include agency oversight costs and all required materials and labor purchased on the open market.

Similar completed projects provide an excellent source of cost information since they reflect real-world experiences. An alternative source of information is the cost estimates from proposed projects that have not yet been implemented, or partially completed projects. In either case, factors that can affect per unit restoration costs, such as fixed costs (e.g., administration, permitting) or donated services and materials, should be accounted for by carefully examining the available cost information. The cost analysis of each restoration alternative should also include the costs for an effective program to monitor the increases in species production. Where costs are not developed on a per unit restored basis, total costs can be divided by the scale of the project to develop the required unit costs. Finally, unit costs are converted to their present value equivalents to simplify addressing costs that may be incurred over a number of years.

3.4.8 Develop total value estimates for I&E losses

After determining the required scale for restoration and the associated unit costs, the eighth step estimates the total value of all I&E losses. The costs associated with a single restoration alternative are determined by multiplying the required scale of implementation to offset I&E losses by the unit cost for the restoration alternative. The total cost of offsetting the I&E losses is then determined by summing the costs of each restoration alternative implemented, following their prioritization for each species.

The total estimated cost of replacing all of the organisms lost is a discrete, present value representing the current cost for providing a stream of increased production benefits for the affected species in perpetuity. In other words, the HRC valuation estimate reflects the cost now for increasing the production of I&E species at an average annual level that would offset the losses in the current year and all future years, all else being equal.

3.5 Strengths and Weaknesses of the HRC

The primary strength of the HRC method is the explicit recognition that I&E losses have impacts on the aquatic ecosystem and the public's use and enjoyment of that ecosystem beyond that estimated by reduced commercial and recreational catches. The HRC method provides a supplemental or alternative option for determining the value of I&E losses of all species, including forage species overlooked by conventional methods, so that the public (i.e., those directly and indirectly affected by I&E), and the regulators who represent them can have greater confidence in the true range of values associated with I&E losses. The need to provide detailed restoration alternatives for the HRC method provides permitting agencies with a means of scaling the mitigation level to offset residual I&E losses associated with a permitted technology. Finally, the HRC method has a strong intuitive appeal as a valuation tool because it uses the costs associated with enhancing natural habitats so that they will produce the equivalent number and type of resources necessary to offset the I&E losses produced by the CWIS.

Public confidence levels associated with the results of an HRC valuation will be determined by the quality of the input data for identifying preferred restoration alternatives, estimating increased production of species following restoration, and deriving appropriate and complete unit costs for restoration alternatives. In this sense, HRC is primarily limited by data quality, rather than any methodological weakness. However, data quality affects HRC and other benefit analyses, alike. EPA's studies are limited by the quality and extent of the impingement and entrainment data collected by the facility. This weakness can be addressed in future analyses by using appropriate guidelines for monitoring I&E, and by planning a more active program of defining expected production increases for species following implementation of different restoration activities. In practice, implementing appropriate monitoring programs for both the harm done by a CWIS and the benefits gained from restoration projects will produce a more comprehensive database. This comprehensive database will then facilitate scaling restoration projects to replace I&E losses. By

ensuring that the costs associated with such monitoring programs are incorporated in the unit costs used to value I&E losses, the HRC method will help develop the information needed to address its primary limitation.

2.2 Strengths and Weaknesses of the HRC

The primary strength of the HRC method is the explicit recognition that I&E losses are not independent on the aquatic ecosystem and the public's use and enjoyment of the ecosystem. I&E losses are estimated by reduced consumption and recreational catches. The HRC method produces a replacement or substitute value for determining the value of I&E losses of all species, including those species considered as commercial methods, so that the public can have greater directly and indirectly affected by I&E, and the regulation of a resource than can have greater confidence in the true value of a loss associated with I&E losses. The need to provide detailed restoration alternatives for the HRC method provides permitting agencies with a means of scaling the restoration level to effect desired I&E losses associated with a particular technology. Finally, the HRC method has a strong financial appeal as a restoration tool because it uses the costs associated with restoring natural habitat so that they will produce the equivalent number and type of resources necessary to offset the I&E losses produced by the activity.

Public confidence levels associated with the results of an HRC restoration will be determined by the quality of the input data for identifying restored restoration alternatives, estimating increased production of species following restoration, and deriving appropriate and complete unit costs for restoration alternatives. In the case of HRC, a primary limitation is data quality, rather than any methodological weakness. However, data quality affects HRC and other benefit transfer, alike. EPA's studies are limited by the quantity and extent of the impairment and management data collected by the facility. This weak area can be addressed to future studies by using appropriate guidelines for monitoring I&E, and by planning a more comprehensive program of habitat restoration. In production fisheries for species following application of different restoration activities, the practice of implementing appropriate monitoring programs for both the natural and the restored habitat is critical. The practice of implementing appropriate monitoring programs for both the natural and the restored habitat is critical. The practice of implementing appropriate monitoring programs for both the natural and the restored habitat is critical.

4. Application of the HRC Method to the Pilgrim Facility

Application of the HRC method to the Pilgrim facility was based on published data wherever possible. Where published data were unavailable or insufficient to address HRC needs, unpublished data from knowledgeable resource experts were used. In some cases, the best professional judgement of these experts was used to apply reasonable assumptions to their data. In these cases, the authors sought ranges beyond which the experts became skeptical, and then applied a conservative (leading to lower restoration costs) assumption from within that range. In other words, this HRC seeks the cost of the minimum amount of restoration necessary to offset I&E losses at the Pilgrim facility, in the opinion of knowledgeable resource experts. Conservative assumptions are identified throughout Chapter 4.

4.1 Step 1: Quantify I&E Losses

Overview of procedure for evaluating I&E

Losses of aquatic resources resulting from I&E were expressed as foregone age-1 equivalents for each species and life stage for which monitoring data are available (Ricker, 1975; Hilborn and Walters, 1992; Quinn and Deriso, 1999). These estimates were developed in conjunction with case studies developed by the Agency as part of the national Section 316(b) rulemaking. These foregone aquatic resources were modeled using facility-specific I&E rates combined with relevant species life history characteristics such as growth rates, natural mortality rates, and fishing mortality rates. The HRC valuation used the average annual I&E losses calculated for each species to determine the amount of natural habitat required to offset the losses for each species.

4.1.1 Source Data

4.1.1.1 Facility I&E monitoring

The inputs for analyses included the empirical I&E counts reported by the facility. Impingement monitoring involved sampling impingement screens or catchment areas, counting the impinged fish, and extrapolating the count to an annual basis. Impingement enumeration procedures were geared toward the types of fish that were impinged, which are typically larger and older than those that are entrained. Entrainment monitoring typically involved intercepting a small portion of the intake flow at a selected location in the facility, collecting fish by sieving the water sample through nets or other collection devices, counting the collected fish, and extrapolating the counts

to an annual basis. Life stage-specific annual losses were used for assessment of entrainment losses, whereas all fish killed by impingement were assumed to be age 1 at the time of death.

4.1.1.2 Species evaluated

Detailed loss analyses were conducted for each species with significant numbers in facility collections or with special significance (e.g., threatened or endangered status). A small fraction of species that were identified in I&E records were not evaluated because of a lack of life history information. These species were treated as an aggregate, and their I&E rates were expressed as a fraction of the total I&E.

4.1.1.3 Life history data

Life history data included mortality rates, growth rates, fraction of each age class vulnerable to harvest, fishing mortality rates, and natural (nonfishing) mortality rates for each species. Each of these parameters was also stage-specific, with the exception of mortality rates, which are typically constant for fish older than a given catchability threshold.

Life history data were obtained from facility reports, the fisheries literature, and publicly available fisheries databases (e.g., Fishbase). To the extent feasible, region-specific life history data most relevant to local populations near the case study facility were used for each species. A static set of life history parameters was used for all data analyses. No stochastic or dynamic effects such as compensatory mortality or growth or random environmental variation were used. Where no information on survival rates was available for individual life stages, survival rates for an equilibrium population were based on records of lifetime fecundity using the relationship presented in Goodyear (1978):

$$S_{eq} = 2/fa \quad (4-1)$$

where:

S_{eq} = the probability of survival from egg to the expected age of spawning females
 fa = the expected lifetime total egg production

Published fishing mortality rates (F) were assumed to reflect combined mortality due to both commercial and recreational fishing. Basic fishery science relationships (Ricker, 1975) among mortality and survival rates were assumed, such as:

$$Z = M + F$$

(4-2)

where:

- Z = the total instantaneous mortality rate
- M = natural (nonfishing) instantaneous mortality rate
- F = fishing instantaneous mortality rate

and

$$S = e^{(-Z)}$$

(4-3)

where:

- S = the survival rate as a fraction.

4.1.2 Biological Models Used to Evaluate I&E

4.1.2.1 Modeling age-1 equivalents

The Equivalent Adult Model (EAM) is a method for expressing I&E losses as an equivalent number of individuals at some other life stage, referred to as the age of equivalency (Horst 1975; Goodyear, 1978; EPRI, 1999). The age of equivalency can be any life stage of interest. The method provides a convenient means of converting losses of fish eggs and larvae into units of individual fish and provides a standard metric for comparing losses among species, years, and facilities. For the Pilgrim HRC valuation, I&E losses were expressed as an equivalent number of age-1 individuals. This is the number of impinged and entrained individuals that would otherwise have survived to be age 1 plus the number of impinged individuals (which are assumed to be impinged at age 1).

The EAM calculation requires life-stage-specific entrainment counts and life-stage-specific mortality rates from the life stage of entrainment to the life stage of equivalence. The cumulative survival rate from age at entrainment until age 1 is the product of all stage-specific survival rates to age 1. The calculation is:

$$S_{j,1} = S_j^* \prod_{i=j+1}^{j_{\max}} S_i \quad (4-4)$$

where:

- $S_{j,1}$ = cumulative survival from stage j until age 1
- S_j = survival fraction from stage j to stage $j + 1$
- S_j^* = $2S_j e^{-\log(1+S_j)}$ = adjusted S_j
- j_{\max} = the stage immediately before age 1.

Equation 4-4 defines $S_{j,1}$, which is the expected cumulative survival rate (as a fraction) from the stage at which entrainment occurs, j , through age 1. The components of Equation 4-4 represent survival rates during the different life stages between life stage j , when a fish is entrained, and age 1. Survival through the stage at which entrainment occurs, j , is treated as a special case because the amount of time spent in that stage before entrainment is unknown, and therefore the known stage-specific survival rate, S_j , does not apply because S_j describes the survival rate through the entire length of time that a fish is in stage j . Therefore, to find the expected survival rate from the day that a fish was entrained until the time that it would have passed into the subsequent stage, an adjustment to S_j is required. The adjusted rate S_j^* describes the effective survival rate for the group of fish entrained at stage j , considering the fact that the individual fish were entrained at various specific ages within stage j .

Age-1 equivalents are then calculated as:

$$AE1_{j,k} = L_{j,k} S_{j,1} \quad (4-5)$$

where:

- $AE1_{j,k}$ = the number of age-1 equivalents killed during life stage j in year k
- $L_{j,k}$ = the number of individuals killed during life stage j in year k
- $S_{j,1}$ = the cumulative survival rate for individuals passing from life stage j to age 1 (Eq. 4-4).

The total number of age-1 equivalents derived from losses at all stages in year k is then given by:

$$AE1_k = \sum_{j=j_{\min}}^{j_{\max}} AE1_{j,k} \quad (4-6)$$

where:

- $AE1_k$ = the total number of age-1 equivalents derived from losses at all stages in year k .

These calculations were used to derive the total age-1 equivalents for each species and year of sampling at Pilgrim.

4.1.2.2 Uncertainty

The modeling methods, assumptions, and results followed sound scientific practice throughout, but it is impossible to avoid uncertainty that may cause the reported results to be imprecise or to carry potential statistical bias. Uncertainty of this nature is not unique to studies of I&E effects (Finkel, 1990).

The analyses attempt to model a process that is enormously complex. The analyses are an interdisciplinary process that spans several major fields of study, including aquatic and marine ecology, fishery science, estuarine hydrodynamics, economics, and engineering, each of which acknowledges its own complex suite of interacting factors. A formal quantification of variability and uncertainty (which could be accomplished by analytic means or by Monte Carlo methods) would require information about the variance associated with each part of this large set of factors, but much of that information is lacking. Because estimates of confidence limits are themselves subject to substantial uncertainty, numeric confidence limits are not reported for these results. Nonetheless, because care was taken to use the best biological models and data available for its I&E evaluations and economic analyses, these results provide a reliable, scientifically sound basis for estimating the potential benefits of minimizing I&E. The models used are based on standard fisheries methods. The I&E data were developed by the industry, and any measurement errors or other uncertainties are beyond control.

The following discussion outlines major uncertainties in these analyses. Uncertainty may be classified into two general types (Finkel, 1990). One type, referred to as structural uncertainty, reflects the limits of the conceptual formulation of a model and relationships among model parameters. The other general type is parameter uncertainty, which flows from uncertainty about any and all of the specific numeric values of model parameters. The following discussion considers these two types of uncertainty in relation to the models used to evaluate I&E.

Structural uncertainty

The models used to assess the consequences of I&E simplify a very complex process. The degree of simplification is substantial but necessary because of the limited availability of empirical data. Table 4-1 provides examples of some potentially important considerations that are not captured by the models. These structural uncertainties should generally lead to inaccuracies, rather than imprecision, in the final results.

Table 4-1. Factors affecting model uncertainty in EPA's assessment of I&E consequences.

Type	General treatment in model	Specific treatment in model
Generally simple structure	Each species lost to I&E treated independently	Fish species considered to fall into several types: harvested (commercial, recreational, or both) or not harvested (forage)
Biological submodels	No dynamic elements	Life history parameters were static (i.e., growth and survival did not vary through time in response to long term trends in community); growth and survival rates in the subpopulation of fish that did not suffer I&E mortality did not change in response to possible compensatory effects

Parameter uncertainty

The models used to evaluate I&E require knowledge of growth rates and mortality rates that vary by species and are often age-specific as well. Uncertainty about the values of these parameters arises for two general reasons. The first source of uncertainty is imperfect precision and accuracy of the original estimate because of unavoidable sampling and measurement errors. The second major source of uncertainty is the applicability of previous parameter estimates to the current situation. Although published parameter estimates were judged to be most pertinent to the region considered, it is unlikely that growth and survival rates would be exactly the same as survival rates developed in a different setting. The applicability of published parameter estimates may also vary through time because of changes in the local ecosystem as a whole, or because of climatological changes and other stochastic factors. All of these types of temporal changes could be manifest as significant temporary effects, or as persistent long-term trends.

Table 4-2 presents some examples of parameter uncertainty. In all these cases, increasing uncertainty about specific parameters implies increasing uncertainty about the reported point estimates of I&E losses. The point estimates are biased only insofar as the input parameters are biased in aggregate (i.e., inaccuracies in multiple parameter values that are above the "actual" values but below the "actual" values in other cases may tend to counteract). In this context, parameter uncertainty should generally lead to imprecision, rather than biases, in the final results.

Uncertainties related to engineering

The evaluation of I&E consequences was also affected by uncertainty about the engineering and operating characteristics of the case study facilities. It is unlikely that plant operating characteristics (e.g., seasonal, diurnal, or intermittent changes in intake water flow rates) were constant throughout any particular year, which therefore introduces the possibility of bias in the loss rates reported by the facilities. The facilities' loss estimates were assumed not to include any intentional biases, omissions, or other kinds of misrepresentations.

Table 4-2. Parameters included in the I&E assessment model that are subject to uncertainty.

Type	Factors	Examples of uncertainties in model
Monitoring/ loss rate estimates	Sampling regimes	Sampling regimes subject to numerous plant-specific difficulties; no established guidelines or performance standards for how to design and conduct sampling regimes
	Extrapolation assumptions	Extrapolation to annual I&E rates requires numerous assumptions required by monitoring designers and analysts regarding diurnal/seasonal/annual cycles in fish presence and vulnerability and various technical factors (e.g., net collection efficiency; hydrological factors affecting I&E rates)
	Species selection	Facilities responding to variable sets of regulatory demands; flexible interpretation; variations in data availability in resulting time series
	Sensitivity of fish to I&E	Through-plant mortality assumed to be 100%; some back-calculations required in cases where facilities had reported only I&E rates that assumed <100% mortality
Biological/ life history	Natural mortality rates	Used stage-specific natural mortality rates (M) for >10 stages per species
	Growth rates	Simple exponential growth rates or simple size-at-age parameters used
	Geographic considerations	Migration patterns; I&E occurring during spawning runs or larval out-migration? Location of harvestable adults; intermingling with other stocks; If compensation occurs, where and when?

4.1.3 I&E Losses at the Pilgrim Facility

The Pilgrim facility has reported that millions of aquatic organisms have been lost to I&E each year since once-through cooling water systems were put in place. Stratus Consulting evaluated all species known to be impinged and entrained by the Pilgrim facility, including commercial, recreational, and forage fish species, based on information provided in facility I&E monitoring reports (New England Power Company and Marine Research Inc., 1995; PG&E Generating and Marine Research Inc., 1999). Table 4-3 lists these species.

Table 4-3. Aquatic species vulnerable to I&E at the Pilgrim Facility.

Common name	Scientific name
Alewife	<i>Alosa pseudoharengus</i>
American eel	<i>Anguilla rostrata</i>
American plaice	<i>Hippoglossoides platessoides</i>
American sand lance	<i>Ammodytes americanus</i>
Atlantic cod	<i>Gadus morhua</i>
Atlantic herring	<i>Clupea harengus</i>
Atlantic mackerel	<i>Scomber scombrus</i>
Atlantic menhaden	<i>Brevoortia tyrannus</i>
Atlantic moonfish	<i>Selene setapinnis</i>
Atlantic silverside	<i>Menidia menidia</i>

Table 4-3. Aquatic species vulnerable to I&E at the Pilgrim Facility (cont.).

Common name	Scientific name
Atlantic tomcod	<i>Microgadus tomcod</i>
Bay anchovy	<i>Anchoa mitchilli</i>
Black ruff	<i>Centrolophus niger</i>
Black sea bass	<i>Centropristis striata</i>
Blackspotted stickleback	<i>Gasterosteus wheatlandi</i>
Blue mussel	<i>Mytilus edulis</i>
Blueback herring	<i>Alosa aestivalis</i>
Bluefish	<i>Pomatomus saltator</i>
Butterfish	<i>Peprilus triacanthus</i>
Cunner	<i>Tautoglabrus adspersus</i>
Flying gurnard	<i>Dactylopterus volitans</i>
Fourbeard rockling	<i>Enchelyopus cimbrius</i>
Fourspot flounder	<i>Paralichthys oblongus</i>
Grubby	<i>Myoxocephalus aeneus</i>
Hake species	<i>Urophycis spp.</i>
Hogchoker	<i>Trinectes maculatus</i>
Little skate	<i>Leucoraja erinacea</i>
Longhorn sculpin	<i>Myoxocephalus octodecemspinosus</i>
Lumpfish	<i>Cyclopterus lumpus</i>
Mummichog	<i>Fundulus heteroclitus</i>
Northern kingfish	<i>Menticirrhus saxatilis</i>
Northern pipefish	<i>Syngnathus fuscus</i>
Northern puffer	<i>Sphoeroides maculatus</i>
Northern searobin	<i>Prionotus carolinus</i>
Orange filefish	<i>Aluterus schoepfii</i>
Pearlside	<i>Maurollicus muelleri</i>
Planehead filefish	<i>Stephanolepis hispidus</i>
Pollock	<i>Pollachius pollachius</i>
Radiated shanny	<i>Ulvaria subbifurcata</i>
Rainbow smelt	<i>Osmerus mordax</i>
Red hake	<i>Urophycis chuss</i>
Rock gunnel	<i>Pholis gunnellus</i>
Round scad	<i>Decapterus punctatus</i>
Sand lance species	<i>Ammodyte spp.</i>
Sculpin species	<i>Cottidae</i>
Scup	<i>Stenotomus chrysops</i>
Searobin	<i>Triglidae</i>
Shorthorn sculpin	<i>Myoxocephalus scorpius</i>

Table 4-3. Aquatic species vulnerable to I&E at the Pilgrim Facility (cont.).

Common name	Scientific name
Silver hake	<i>Merluccius bilinearis</i>
Silver rag	<i>Ariomma bondi</i>
Smallmouth flounder	<i>Etropus microstomus</i>
Smooth dogfish	<i>Mustelus canis</i>
Snailfish species	<i>Cyclopteridae</i>
Spiny dogfish	<i>Squalus acanthias</i>
Spot	<i>Leiostomus xanthurus</i>
Spotted hake	<i>Urophycis regia</i>
Striped bass	<i>Morone saxatilis</i>
Striped cusk-eel	<i>Ophidion marginatum</i>
Striped killifish	<i>Fundulus majalis</i>
Striped searobin	<i>Prionotus evolans</i>
Summer flounder	<i>Paralichthys dentatus</i>
Tautog	<i>Tautoga onitis</i>
Threespine stickleback	<i>Gasterosteus aculeatus</i>
White hake	<i>Urophycis tenuis</i>
White perch	<i>Morone americana</i>
Windowpane	<i>Scophthalmus aquosus</i>
Winter flounder	<i>Pleuronectes americanus</i>
Yellowtail flounder	<i>Limanda ferruginea</i>

Sources: Stone & Webster Engineering Corporation, 1977; Boston Edison Company, 1991-1994, 1995a, 1995b, 1996-1999.

Of the 63 species, 2 genera, and 3 families of fish listed in Table 4-3, the 34 taxa that had losses greater than 0.1% of the total impingement or total entrainment losses at the facility (the criterion for inclusion in the EAM) were incorporated into the HRC analysis. The average annual age-1 equivalent losses to impingement and entrainment at Pilgrim for these 34 taxa over the 1973 through 1999 period are presented in Table 4-4, in order of decreasing mean annual I&E losses.

Table 4-4. Age-1 equivalent I&E losses of fishes at the Pilgrim Facility.

Species	Mean annual age-1 equivalent impingement 1974-1999	Mean annual age-1 equivalent entrainment 1974-1999	Total of mean annual I&E (age-1 equivalents)
Finfish			
Rock gunnel	77	4,862,795	4,862,872
American sand lance	27	4,116,258	4,116,285
Radiated shanny	54	1,644,402	1,644,456
Rainbow smelt	6,885	1,323,137	1,330,022
Cunner	411	993,500	993,911

Table 4-4. Age-1 equivalent I&E losses of fishes at the Pilgrim Facility (cont.).

Species	Mean annual age-1 equivalent impingement 1974-1999	Mean annual age-1 equivalent entrainment 1974-1999	Total of mean annual I&E (age-1 equivalents)
Sculpin spp.	13	734,760	734,773
Fourbeard rockling	2	411,189	411,191
Winter flounder	1,144	209,571	210,715
Atlantic herring	8,836	20,243	29,079
Atlantic silverside	20,842	5,087	25,929
Windowpane	284	17,258	17,542
Atlantic menhaden	6,165	8,105	14,270
Atlantic mackerel	3	6,659	6,662
Alewife	4,343	-	4,343
Searobin	69	3,698	3,767
Atlantic cod	301	2,138	2,439
Red hake	229	1,545	1,774
Lumpfish	217	1,080	1,297
Tautog	201	875	1,076
Grubby	879	NA	879
Blueback herring	703	NA	703
Pollock	33	492	525
Butterfish	399	NA	399
American plaice	-	221	221
Northern pipefish	118	NA	118
Threespine stickleback	118	NA	118
Scup	114	NA	114
Striped killifish	90	NA	90
Little skate	78	NA	78
White perch	73	NA	73
Bay anchovy	18	NA	18
Striped bass	9	NA	9
Bluefish	2	NA	2
Hogchoker	2	NA	2
Total age-1 equivalent finfish losses	52,739	14,363,013	14,415,752
Shellfish			
Blue mussel	1.5E+1	1.60E+11	1.60E+11
Total age-1 equivalent shellfish losses	1.5E+1	1.60E+11	1.60E+11

Source: U.S. EPA calculations of age-1 equivalents from I&E data in annual biological monitoring reports by the Pilgrim facility. Details of these calculations are presented in a benefits case study for the 316(b) rulemaking, available from the Office of Science and Technology, Office of Water, U.S. EPA, Washington, DC.

In addition, quantitative estimates of blue mussel losses were available for a number of years in Pilgrim's I&E monitoring reports. The losses for blue mussels were quantified as age-1 equivalents using the same EAM model. The I&E losses for blue mussels are also presented in Table 4-4.

4.2 Step 2: Identify Habitat Requirements

Determining the best course of action for restoring habitat to offset losses of species to I&E requires understanding the specific habitat requirements for each species. Habitat requirements for fish may include physical habitat needs such as substrate types and geographic locations as well as water quality needs and food sources. This section gives a detailed summary of the habitat components needed for the critical lifestages of species that are lost as a result of I&E.

Physical habitat requirements for 34 identified species

Alewife (*Alosa pseudoharengus*)



Source: New York Sportfishing
and Aquatic Resources
Educational Program, 2001

The alewife is a member of the Clupeidae (herring) family. Alewife ranges along the western Atlantic coast from Newfoundland to North Carolina (Scott and Crossman, 1973), and tends to be more abundant in the mid-Atlantic and along the northeastern coast. It is also found in the St. Lawrence River and the Great Lakes. Alewife are anadromous, migrating inland from coastal waters in the spring to spawn. Adult alewife overwinter along the northern continental shelf, settling at the bottom in depths of 56 to 100 m (Able and Fahay, 1998).

Spawning takes place in the upper reaches of coastal rivers, in slow-flowing sections of slightly brackish or fresh water. Spawning is temperature-driven, beginning in the spring as water temperatures reach 13 to 15°C, and ending when temperatures exceed 27°C (Able and Fahay, 1998). Females lay demersal eggs in shallow water less than 2 m deep.

Larvae remain in the upstream spawning area for some time before drifting downstream to natal estuarine waters. Juveniles exhibit a diurnal vertical migration, remaining near the bottom during the day and rising to the surface at night (Waterfield, 1995). In the fall, juveniles move offshore to nursery areas (Able and Fahay, 1998).

Ecologically, alewife is an important prey item for many fish (including striped bass, weakfish and rainbow trout), and commercial landings of alewife have ranged from a high of 34 million kg in 1958 to a low of less than 3 million kg in recent years (ASMFC, 2000). Alewife has been introduced to a number of lakes to provide forage for sport fish (Jude et al., 1987).

American plaice (*Hippoglossoides platessoides*)



Source: Newfoundland and Labrador Fisheries and Aquaculture, 2001

The American plaice is a member of the Pleuronectidae (one of the flounder families) family. Its geographic distribution extends from Labrador, Canada, south to Cape Cod, Massachusetts. It is also present on the eastern side of the Atlantic along the coast of Europe (Bigelow and Schroeder, 1953). It is the most abundant of the flatfish species in the northwest Atlantic (Johnson et al., 1999a). As the abundance of other flatfish species has decreased, the commercial importance of American plaice has grown (Johnson et al., 1999a).

Spawning occurs from March until the middle of June in waters north of Cape Cod (Bigelow and Schroeder, 1953). Females may lay 50,000 to 3 million eggs within their lifetime (Froese and Pauly, 2000). Spawning occurs at depths less than 90 m as adults migrate to shallower waters. The buoyant eggs are released near the bottom of the water column and drift to the upper water column (Johnson et al., 1999a).

Larvae hatch out at approximately 2.4 mm (Johnson et al., 1999a). Larval stage American plaice range in size from 5.1 to 16.4 cm (Johnson et al., 1999a). Larvae have been found at depths ranging from 30 to 210 m, with the highest abundance at 50 to 90 m (Johnson et al., 1999a). During the larval stage, the left eye migrates to the right side of the fish as the fish matures and flattens out. By the first winter, juvenile American plaice can reach 7.6 cm (Johnson et al., 1999a).

Sexual maturity begins at 2 to 3 years, and all individuals are mature by age 4. American plaice have been documented to live up to 30 years and reach lengths of up to 82 cm (Froese and Pauly, 2000).

American sand lance (*Ammodytes americanus*)



Source: Annenberg/CPB, 2001

The American sand lance is a member of the family Ammodytidae (sand lances). It is a small, bottom-dwelling species that ranges from Labrador, Canada, to Delaware Bay (Able and Fahay, 1998). When they are not schooling, they bury themselves in sand with only their heads emerging (Scott and Scott, 1988). American sand lances are typically found in protected bays and estuaries and in shallow coastal waters (Froese and Pauly, 2000).

Within the range of Nova Scotia to Long Island, spawning occurs from December to January (Scott and Scott, 1988). Spawning is thought to occur over sandy bottoms (Bigelow and Schroeder, 1953). Females may release from 1,855 to 5,196 eggs, with a reported average of 3,475 eggs. Eggs are 0.67 to 1.01 mm in diameter. Larvae hatch out at approximately 4 mm. The habits of young-of-the-year are not well known.

Atlantic cod (*Gadus morhua*)



Source: Maine Division of Marine Resources, 2001

Atlantic cod is a member of the Gadidae family, which includes cods, hake, and haddocks. The species is found from Greenland south to Cape Hatteras, North Carolina (Fahay et al., 1999a). Adult Atlantic cod live in diverse habitats ranging from inshore waters to the outer continental shelf, and from depths of over 400 m to surface waters. They generally prefer cooler water temperatures of -0.5 to 10°C (Scott and Scott, 1988). Off the New England coast, Atlantic cod migrate seasonally, moving into coastal waters in the fall and returning to deeper waters during spring (Fahay et al., 1999a).

Spawning begins in northern areas as early as February and ends in southern areas as late as December (Scott and Scott, 1988). Cod spawn repeatedly for up to 50 days once a year (Kjesbu, 1989). Spawning occurs at depths from less than 110 m to more than 182 m, depending on water temperature. Eggs are distributed throughout the water column, although their buoyancy tends to concentrate them in a cold intermediate layer if the water is stratified (Fahay et al., 1999a).

The pelagic larvae move to the bottom during the day and rise at night (Lough and Potter, 1993; Gotceitas et al., 1997). Both age 0 and age-1 cod are found in nearshore environments, preferably over sandy substrates (Fraser et al., 1996), and young cod often seek cover in eelgrass (Gotceitas et al., 1997). Juveniles 40 mm or larger are demersal, but will rise up to 5 m off the bottom at night (Lough and Potter, 1993).

Atlantic herring (*Clupea harengus*)



Source: NOAA, 2001b

The Atlantic herring is a member of the family Clupeidae. Atlantic herring range from southwestern Greenland and Labrador to South Carolina (Scott and Scott, 1988). Adults are found in coastal and continental shelf waters at depths of up to 200 m (656 ft) and in water temperatures from 1 to 18°C (ASMFC, 2001; Froese and Pauly, 2000). Feeding migrations may consist of hundreds of thousands of adults. Schools are composed of individuals of similar size classes, and tend to inhabit the upper water column. Most Atlantic herring migrate south in the fall from feeding grounds off Maine to southern New England (Kelly and Moring, 1986).

Spawning occurs throughout the year, peaking in shallow waters in the spring and deeper waters in the fall. Adults may travel long distances to return to spawning grounds (Kelly and Moring, 1986). Spawning habitat consists of rock, gravel, or sandy substrates 15 to 45 m deep. Atlantic herring eggs are demersal, stick to the bottom in clumps or layers, and often cover the substrate (ASMFC, 2001).

Larvae disperse to estuaries after hatching, and grow to approximately 30 mm long before transforming into juveniles (Able and Fahay, 1998). Transformation occurs after about 152 days at 7 to 12°C (Doyle, 1977). Larvae hatched earlier in the season tend to grow faster than those hatched later (Jones, 1985). These juveniles move in large inshore schools.

Herring fisheries developed in the late 1800s concurrent with the development of canning technology. Herring were also used as bait for the lobster industry, which developed at about the same time. Annual landings were as high as 68 million kg in the late 1800s (ASMFC, 2001). Overfishing, particularly aggressive foreign fisheries that developed in the 1960s on Georges Bank with landings peaking at 363 million kg in 1968, contributed to a crash of the Atlantic herring population. Current annual harvests are in the range of 36 to 45 million kg. Primary uses of Atlantic herring are as canned sardines, steaks, and bait for crab, lobster, and tuna fisheries (ASMFC, 2001). Larger juveniles are referred to as "sardines" and are harvested commercially (Jury et al., 1994)

Atlantic mackerel (*Scomber scombrus*)



Source: NOAA, 2001b

Atlantic mackerel is a member of the Scombridae family, which includes mackerels, tunas, and bonitos. Atlantic mackerel range from Labrador to Cape Lookout, North Carolina. The species tends to school in large groups in shelf areas with water temperatures of 9 to 12°C (Scott and Scott, 1988).

Winters are spent in deeper waters, but mackerel return to shore in springtime to spawn. There are two major spawning areas for Atlantic mackerel: between Cape Cod and Cape Hatteras, and in the Gulf of St. Lawrence (Scott and Scott, 1988). In the northern regions of the Mid-Atlantic Bight they spawn from April to June (Ware and Lambert, 1985). In summer and fall, fish from the Mid-Atlantic Bight move into coastal areas along the Gulf of Maine, while the northern contingent remains in Canadian waters (Ware and Lambert, 1985).

Eggs are pelagic and are released near the surface, in the upper 15 m of water. After spawning, adults generally migrate in schools to offshore feeding areas before returning to their overwintering sites (Scott and Scott, 1988). Once juveniles join the offshore adults, they remain in schools. Adults are obligate swimmers because of the absence of a swim bladder (Scott and Scott, 1988).

Atlantic mackerel is fished both commercially and for sport. Fish caught in the United States and Canada peaked in 1973 at 400,000 tons per year and declined to a low of 30,000 tons in the late 1970s. Weak year classes occurred from 1975 through 1980, but stocks have been very high (Anderson, 1995).

Atlantic menhaden (*Brevoortia tyrannus*)

Source: NOAA, 2001c

The Atlantic menhaden is a member of the Clupeidae family, and is a euryhaline species, occupying coastal and estuarine habitats. It is found along the Atlantic coast of North America, from Maine to northern Florida (Hall, 1995). Adults congregate in large schools in coastal areas; these schools are especially abundant in and near major estuaries and bays.

Atlantic menhaden spawn year round at sea and in larger bays (Scott and Scott, 1988). Spawning peaks during the southward fall migration and continues throughout the winter off the North Carolina coast. There is limited spawning during the northward migration and during the summer (Hall, 1995). The majority of spawning occurs over the inner continental shelf, with lesser activity in bays and estuaries (Able and Fahay, 1998).

Females mature just before age 3, and release buoyant, planktonic eggs during spawning (Hall, 1995). Atlantic menhaden annual egg production range from approximately 100,000 to 600,000 eggs for fish age 1 to age 5 (Dietrich, 1979).

Larvae hatch after approximately 24 hours and remain in the plankton. Those larvae that hatch at sea enter estuarine waters 1 to 2 months later (Hall, 1995). Water temperatures below 3°C kill the larvae, and therefore larvae that fail to reach estuaries before the fall are more likely to die than those arriving in early spring (Able and Fahay, 1998).

During the fall and early winter, most menhaden migrate south to the North Carolina capes, where they remain until March and early April. They avoid waters below 3°C, but can tolerate a wide range of salinities from less than 1‰ up to 33-37‰ (Hall, 1995). Sexual maturity begins just before age 3 (Hall, 1995), and menhaden return to the shelf waters of southern New England to spawn in the summer. Menhaden also spawn in early spring and winter off North Carolina and in spring and late fall in the mid-Atlantic region (Wang and Kernehan, 1979). However, primary spawning grounds for Atlantic menhaden are offshore near Cape Cod (Jury et al., 1994).

Atlantic silverside (*Menidia menidia*)

Source: Maryland DNR, 2001

The Atlantic silverside is a member of the Atherinidae (silversides) family. Its geographic range extends from the coastal waters of New Brunswick to northern Florida (Fay et al., 1983a), but it is most abundant between Cape Cod and South Carolina (Able and Fahay, 1998). Silversides prefer moderately saline estuarine areas and sandy or gravelly habitats (U.S. EPA, 1982), sand bars, open beaches, tidal creeks, river mouths, and flooded vegetation zones (Fay et al., 1983a).

Atlantic silversides spawn in the upper intertidal zone during spring and summer. Spawning appears to be stimulated by new and full moons, in association with spring tides. During the summer, juveniles occupy estuaries, including intertidal creeks, marshes, and shore zones of bays

and estuaries. Silversides typically migrate offshore in the winter (McBride, 1995). In studies of seasonal distribution in Massachusetts, all individuals left inshore waters during winter months (Able and Fahay, 1998).

Bay anchovy (*Anchoa mitchilli*)



Source: NOAA Coastal Service Center, 2001

The bay anchovy is a member of the Engraulidae (anchovy) family, and is one of the most abundant species in estuaries along the Atlantic and Gulf coasts of the United States (Voughlitois et al., 1987). Because of its widespread distribution and overall abundance, bay anchovy are an important component of the food chain (Morton, 1989).

Bay anchovy is a pelagic species commonly found in shallow tidal areas with muddy bottoms and brackish waters (Froese and Pauly, 2000). It tends to be found in higher densities in vegetated areas such as eelgrass beds (Castro and Cowen, 1991).

The spawning period of bay anchovy is long, with records ranging from April to November (Voughlitois et al., 1987). Spawning occurs over a wide range of salinities, but has been correlated with areas of high zooplankton abundance and low abundance of predators (Able and Fahay, 1998). The eggs are pelagic, and the survival rate of the eggs may decrease with increases in water salinity (Dovel, 1971). Most young-of-year migrate out of the estuaries at the end of the summer in schools, and can be found in large numbers on the inner continental shelf in the fall (Voughlitois et al., 1987).

Blueback herring (*Alosa aestivalis*)



Source: New York Sportfishing and Aquatic Resources Educational Program, 2001

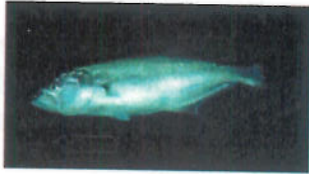
The blueback herring is a member of the Clupeidae family. The range of blueback herring extends from Nova Scotia south to northern Florida, though they are more abundant in the southern portion of their range (Scott and Scott, 1988).

Adults spawn from spring to early summer in upstream brackish or freshwater areas of rivers and tributaries. Spawning occurs at night in fast currents over a hard substrate (Loesch and Lund, 1977). Spawning groups have been observed diving to the bottom and releasing the semi-adhesive eggs over the substrate, but many eggs are dislodged by the current and enter the water column. After spawning, adults move downstream and return to the ocean. Over half of the adults are repeat spawners, returning to natal spawning grounds every year (Scherer, 1972).

Eggs float near the bottom for 2 to 4 days, depending on temperature, until hatching (Jones et al., 1978). Juveniles are distributed high in the water column and avoid bottom depths (Able and Fahay, 1998). In the early juvenile stages, fish are swept downstream by the tide. Some juveniles will move upstream until late summer before migrating downstream in late summer to early fall. Juveniles are sensitive to sudden water temperature changes, and emigrate downstream in

response to a decline in temperature (Able and Fahay 1998). By late fall, most young-of-year emigrate to ocean waters to overwinter (Wang and Kernehan, 1979).

Bluefish (*Pomatomus saltatrix*)



Source: Froese and Pauly, 2000

The bluefish is a member of the family Pomatomidae. It is a widely distributed species and can be found in temperate and tropical waters along the continental shelf and in estuarine habitats from Nova Scotia south to Mexico. It is also found along the coasts of Australia, parts of South America, and Africa, and in the Mediterranean Sea (Pottern et al., 1989). There are several recognized geographical races.

Bluefish are most common along surf beaches and rock headlands in clean, high energy waters. Adults can also be found in estuaries and brackish water. They tend to travel in loose schools, feeding voraciously on other fish and killing more than they eat (Froese and Pauly, 2000). They are often associated with sharks and billfish. Adults will migrate to warmer water during winter and to cooler water in summer (Froese and Pauly, 2000).

Bluefish are thought to be serial spawners. The first major spawning event occurs in the Southern Atlantic Bight from March to May. A second major spawning occurs in the Mid-Atlantic Bight from June to August. While these spawning events were previously thought to be the result of two separate spawning populations, there is now evidence of a single, migratory spawning population (Fahay et al., 1999b). Eggs and sperm are broadcast in ocean waters. The buoyant eggs are 0.9 to 1.2 mm in diameter. Larvae hatch out at 2.0 to 2.4 mm (Pottern et al., 1989) and are pelagic, migrating to the surface at night and remaining at a depth of approximately 4 ft during the daylight (Fahay et al., 1999b).

Young juveniles may enter estuaries during the summer and early fall where they may feed heavily, then migrate south to overwinter south of Cape Hatteras (Pottern et al., 1989; Fahay et al., 1999b). Some juveniles remain at sea.

Blue mussel (*Mytilus edulis*)



Source: Interagency
Committee for Outdoor
Recreation, 2001

The blue mussel is an invertebrate, a mollusc, and a member of the Mytilidae family. It is a widely distributed species, occurring in the Arctic, North Atlantic, and Pacific oceans. Along the western Atlantic, its range extends from Labrador to Cape Hatteras, North Carolina (Newell, 1989). Blue mussels provide habitat and food for valuable fish such as tautog, scup, and black sea bass (Steimle, 1995).

Blue mussels can be found in littoral to shallow sublittoral areas from oceanic to brackish estuarine waters. It has evolved a number of sophisticated adaptations that enable it to survive in a wide range of habitats. It is able to tolerate salinities ranging from 5 to 34 ppt, and can survive

being frozen for 8 months each year, as occurs near Labrador. Blue mussel habitat must have sufficient flow to carry suspended food particles and ensure larval dispersal (Newell, 1989).

Blue mussels require a minimum water temperature of 12/C to spawn (Hawkins, 1994), and fertilization has been found to be unsuccessful at salinities of less than 15 ppt (Hawkins, 1994). Spawning near Woods Hole, Massachusetts, has been reported from early February to the end of August (Hawkins, 1994). Spawning occurs into the overlying water column, and attachment to surfaces is highly dependent on tides and currents (Steimle, 1995). The normal duration of planktonic existence is 3 to 4 weeks, but 10 weeks may elapse before settlement (Hawkins, 1994).

Eggs and larvae are free-floating in the water column. The larval stage lasts from 15 to 35 days, depending on environmental conditions. Younger larvae tend to swim near the surface, and all larvae alter their swimming behavior in response to various environmental stimuli (Newell, 1989). The settlement stage begins with the development of the foot, or pediveliger at approximately 3 to 4 weeks (Newell, 1989).

Blue mussels can attach themselves to almost any firm surface, including other mussels. They are often found on rock or coarse gravel, but may colonize in mud and sand substrates if they can find something to attach to.

Butterfish (*Peprilus triacanthus*)



Source: Victorian
Recreational Fishing
Guide, 2001

The butterfish is a member of the family Stromateidae. The butterfish is found along the Atlantic coast from Newfoundland to Florida. They occur in marine and brackish water from the continental shelf (up to 420 m depth) to inshore areas, including the surf zone. They are common in bays and estuaries and are usually found in schools over sand, silty sand, or muddy bottoms. In Narragansett Bay, butterfish have been collected in every season, but are most abundant in the summer (Cross et al., 1999).

During the summer, butterfish move north and inshore to feed and spawn. Butterfish are known to spawn anywhere from coastal bay estuaries to a few miles out to sea. Eggs and larvae are pelagic. Small juveniles often congregate under floating objects, or under jellyfishes. In the winter, they move south and offshore (Cross et al., 1999).

Cunner (*Tautoglabrus adspersus*)



Source: Maine Division of
Marine Resources, 2001

The cunner is a member of the family Labridae (wrasses). The cunner is a dominant component of many temperate marine communities of the western Atlantic Ocean from Newfoundland to Chesapeake Bay (Bigelow and Schroeder, 1953). It is a territorial and sedentary species that occupies small, localized ranges within 10 km of shore. The species prefers complex habitats with natural or artificial structures,

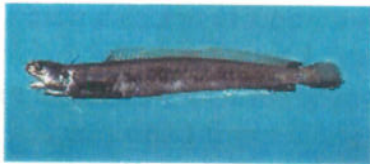
such as bedrock outcrops, glacial boulders, pilings, shipwrecks, or breakwaters, and juveniles inhabit shallow waters (Entergy, 2000).

In Cape Cod Bay, cunner spawn close to shore from mid-March until mid-July (Entergy, 2000). Spawning peaks in waters near Woods Hole, Massachusetts, during the first 3 weeks of June (Entergy, 2000). Males and females are able to spawn several times in a day (Pottle and Green, 1979).

Cunner eggs are pelagic and range from 0.84 to 0.92 mm in diameter (Able and Fahay, 1998). Eggs hatch after several days in water temperatures of 12.8 to 18.3/C (Bigelow and Schroeder, 1953).

Adults do not migrate extensively, but they travel short distances to escape extremes in temperature (Bigelow and Schroeder, 1953). They move to protected areas in the fall and become inactive as water temperatures fall to 7 to 8/C. As temperatures decrease further, cunner become dormant (Olla et al., 1975). Some may overwinter in their summer habitat, but inshore areas that are susceptible to thermal currents are not suitable for the dormant period (Dew, 1976). When spring water temperatures reach 5 to 6/C, cunner move to seasonally transitory habitats such as mussel beds and seaweed (Olla et al., 1979). Cunner are active during the day and become inactive and seek cover at night (Olla et al., 1975).

Fourbeard rockling (*Enchelyopus cimbrius*)



Source: Source: Froese and Pauly, 2000

The fourbeard rockling is a member of the family Gadidae. The fourbeard rockling can be found on both the eastern and the western side of the Atlantic Ocean. Along the coast of North America, it ranges from Newfoundland south to the Gulf of Mexico (Bigelow and Schroeder, 1953).

Fourbeard rocklings are bottom-dwellers, preferring soft bottom such as muddy sand between patches of hard substrate, or the soft bottoms of deep sinks on the continental slopes of both sides of the North Atlantic (Froese and Pauly, 2000). They have been found at a range of depths from a meter along the New England shore to 50 m in the Gulf of St. Lawrence (Bigelow and Schroeder, 1953).

Young larvae are pelagic, and drift at the surface for several months until settling at the bottom. Being at the mercy of the currents, they are sometimes cast ashore (Bigelow and Schroeder, 1953).

Grubby (*Myoxocephalus aeneus*)

Source: Woods Hole
Oceanographic Institution, 2001

The grubby is a member of the Myoxocephalus (sculpins) family. Grubbies occur along the western Atlantic coast from the Gulf of St. Lawrence south to New Jersey (Bigelow and Schroeder, 1953). They inhabit estuaries and coastal waters to depths of up to 130 m deep, and prefer water temperatures between 0 and 21/C (Froese and Pauly, 2000). Grubbies can be found in a wide range of habitats and over many bottom substrates, but they are often found among eelgrass (Bigelow and Schroeder, 1953).

Evidence of spawning has been found in both estuaries and coastal ocean waters. The adhesive eggs sink to the bottom and stick to any surface available. Larvae can be found in a wide range of habitats (Bigelow and Schroeder, 1953).

Hogchoker (*Trinectes maculatus*)

Source: North American Native
Fisheries Association, 2001

The hogchoker is a member of the Achiridae (one of the flounder families) family and is found along the Atlantic coast from Massachusetts to Panama. It is found adjacent to the coast in bays and estuaries, and most frequently in brackish water, although it occasionally runs up into freshwater (Bigelow and Schroeder, 1953).

Spawning occurs in late spring and early summer in brackish waters with an average salinity between 10 and 16 ppt. Eggs are semibuoyant. Larger larvae and juveniles migrate upstream into low salinity nursery areas. Both the young and adults overwinter in the upper parts of estuaries and migrate into higher salinity waters in the spring (Able and Fahay, 1998).

Lumpfish (*Cyclopterus lumpus*)

Source: Newfoundland
and Labrador Fisheries
and Aquaculture, 2001

The lumpfish, or lumpsucker, is a member of the Cyclopteridae (seasnails) family and can be found along the western Atlantic Ocean from Labrador, Canada, south to New Jersey. It is mainly a bottom dweller, found hiding along cold water bottoms, holding onto rocks or other objects with its sucker. It is sometimes found at the surface hiding among seaweed and rockweed. In the Gulf of Maine, lumpfish are found at shallower depths, but they can be found anywhere from tidemark to 550 m along the eastern Atlantic coast. It is a solitary species (Bigelow and Schroeder, 1953).

Eggs and larvae can be found in shallow water along the coastline and at Georges Bank. Egg masses sink to the bottom and often adhere to the surfaces of rocks. The pelagic larvae can be found clinging to seaweed (Bigelow and Schroeder, 1953).

Northern pipefish (*Syngnathus fuscus*)



Source: NOAA, 2001b

The northern pipefish is a member of the Syngnathidae (seahorse) family and is widely distributed, ranging along the western Atlantic coast from the Gulf of St. Lawrence south to northeastern Florida. It can also be found in the Gulf of Mexico. Adults are commonly found in seagrass beds and other submerged vegetation in bays, estuaries, harbors, rivers, creeks, and marshes. In some areas, they exhibit a seasonal migration to deeper oceanic waters along the continental shelf, while in other areas such as near Woods Hole, Massachusetts, they are resident in the eelgrass year round (Bigelow and Schroeder, 1953; Able and Fahay, 1998).

After spawning, males carry the fertilized eggs in a pouch until they hatch. Larvae and juveniles can be found amidst submerged aquatic vegetation in shallow, protected waters. Early life stages of northern pipefish can be found in every estuary along the Mid-Atlantic Bight (Able and Fahay, 1998).

Northern searobin (*Prionotus carolinus*)



Source: NOAA, 2001d

The northern searobin is a member of the Triglidae (searobins) family and occurs along the Atlantic coast from the Gulf of Maine to South Carolina. Depending on the time of year, they occupy habitats ranging from estuaries to the edge of the continental shelf. Between May and October, northern searobins prefer coastal waters with sandy bottom substrates. They overwinter along the continental shelf along the mid- to outer shelf (Able and Fahay, 1998). Juveniles and adults are more common in estuaries of the northern part of their range than in the southern.

Spawning occurs in the summer in estuaries and along the continental shelf throughout the Mid-Atlantic Bight. Larvae are initially pelagic, but soon settle out as they grow. Adults tend to remain near the bottom, preferring hard, smooth substrates (Bigelow and Schroeder, 1953). Eggs, larvae, and juveniles are present in most estuaries of the Mid-Atlantic Bight.

Pollock (*Pollachius virens*)



Source: NOAA, 2001b

The pollock is a member of the Gadidae family. It is present on both the eastern and western coast of the Atlantic Ocean. Along the western coast, it ranges north from Labrador, Canada, south to Cape Hatteras, North Carolina (Scott and Scott, 1988). It can be found at a depth range of approximately 40 to 400 m, but is more common within the range of approximately 100 to 200 m (Scott and Scott, 1988). They can survive in water temperatures as low as 0°C (Bigelow and Schroeder, 1953).

In the northern part of their range, pollock begin spawning in the fall; spawning is most intense in the winter near the Gulf of Maine. Farther south, spawning may occur in the spring (Able and Fahay, 1998). An average female will produce 225,000 eggs, and larger females may produce over 4 million eggs (Bigelow and Schroeder, 1953). Eggs are pelagic, and under typical conditions hatch in approximately 9 days (Scott and Scott, 1988).

Larvae are 3 to 4 mm upon hatching (Able and Fahay, 1998). They are present along the continental shelf from February to May. In February and March, juveniles begin moving inshore and enter inlets and estuaries, where they spend their first year, though some juveniles may remain in marine waters for close to 6 months before moving inshore (Able and Fahay, 1998).

Adults remain in deeper inshore waters, exhibiting migratory patterns during spawning seasons (Scott and Scott, 1988). Sexual maturity is reached for both sexes in the third year (Able and Fahay, 1998). Pollock may live up to 14 years (Scott and Scott, 1988), and may grow up 120 cm, though they rarely exceed 110 cm (Cargnelli et al., 1999).

Radiated shanny (*Ulvaria subbifurcata*)



Source: Woods Hole
Oceanographic Institute, 2001

The radiated shanny is a member of the Stichaeidae, or prickleback, family and is found on the Atlantic coast from northern Newfoundland to southern Massachusetts (Froese and Pauly, 2000). It is a demersal marine species that lives among seaweeds or in rocky interstices. It can also be found over hard bottom in deeper water down to at least 55 m. Adults are inactive during the day, seeking out cover, and feeding during the night and evening (Scott and Scott, 1988).

Spawning occurs from early spring to summer. Eggs are demersal and adhere to each other. Males typically guard and tend several clusters of eggs simultaneously. Eggs hatch in 35 to 40 days when the water temperature reaches 4 to 9°C. Larvae are pelagic until they reach approximately 7mm (Scott and Scott, 1988).

Rainbow smelt (*Osmerus mordax*)



Source: NOAA, 2001b

The rainbow smelt is an anadromous fish belonging to the Osmeridae (smelt) family and ranges from Labrador, Canada, south to the Delaware River. It is also found along the St. Lawrence River and in the Great Lakes. Rainbow smelt are typically found in estuaries or close to the shore at depths less than 6 m. Adults overwinter in estuaries (Buckley, 1989a).

Spawning begins in the spring with smelt running up into freshwater when the water reaches 4 to 9°C. Spawning typically occurs above the head of the tide over gravel substrate in water depths of 0.1 to 1.3 m. Adult smelt spawn at night and return to the estuary during the day. Egg survival rates have been correlated to increasing water currents up to 60 to 80 cm/s. After hatching, eggs

attach to rocks, gravel, or submerged vegetation. After hatching, larvae drift into brackish waters. As the juveniles age, they move to more saline waters (Buckley, 1989a).

Red hake (*Urophycis chuss*)



Source: NOAA, 2001b

The red hake belongs to the Gadidae family and occurs along the western Atlantic Ocean from Nova Scotia south to North Carolina, with a greater abundance between Georges Bank and Hudson Canyon. Adults undergo a seasonal migration, moving inshore in the warmer months and overwintering in deeper waters along the continental shelf. Adults prefer soft mud, silt, or sandy bottoms, but can also be found over rocky bottoms. They occur at depths ranging from shallow bays to 550 m along the continental shelf (Able and Fahay, 1998).

Spawning occurs along the continental shelf. Little is known about the habitat of eggs and young larvae. The pelagic larvae are known to occupy the upper water column from May through December (Steimle et al., 1999).

Pelagic juveniles can be found hiding amongst floating debris, seaweed, and jellyfish. In the fall, demersal settlement occurs and the young juveniles can be found in depressions in the seabed. Older juveniles seek shelter with some form of structure, and are often found amongst sea scallops. Juveniles overwinter in estuaries along the Mid-Atlantic Bight (Steimle et al., 1999).

Rock gunnel (*Pholis gunnellus*)



Source: Froese and Pauly, 2000

The rock gunnel is a member of the Pholidae (gunnels) family and can be found on both sides of the Atlantic Ocean. Along the western Atlantic, rock gunnels range from Labrador south to Delaware Bay, but are not commonly found south of New England. They prefer intertidal habitats such as tidal pools, where they hide under rocks and in crevices (Able and Fahay, 1998). They can remain in shallow intertidal pools for periods of time under rocks or seaweed (Bigelow and Schroeder, 1953), yet have also been found at depths of 183 m on Georges Bank (Able and Fahay, 1998).

Rock gunnel eggs are deposited in masses in a variety of bottom types. Descriptions of nest sites have ranged from empty oyster shells in shallow water to depths of 22 m (Able and Fahay, 1998). Larvae are pelagic up until approximately 30 to 35 mm. Juveniles are cryptic, and little information is known on settlement ecology.

Sculpin species (*Myoxocephalus* spp.)

Source: Froese and Pauly, 2000

The longhorn and shorthorn sculpin belong to the Cottidae family and can be found on both sides of the Atlantic coast and in the Arctic seas. Along the western Atlantic coast, they occur from the Arctic sea south to southern New England. Shorthorn sculpin are cold water fish, and are rarely found as far south as New Jersey, while longhorn sculpin may be commonly found in water near New Jersey and have been reported as far south as Virginia.

Shorthorn sculpin prefer habitats of shallow water with relatively smooth bottoms near ledges or in bays. They tend to stay near the bottom and are sluggish. If disturbed, they tend to not move very far from their original location. Longhorn sculpin occupy a larger range of habitats, from shallow estuary waters to depths of 100 m or more along coastal waters (Bigelow and Schroeder, 1953).

Shorthorn sculpin are able to tolerate colder waters than longhorn. They are able to overwinter in shallow waters, while longhorn sculpin descend into deeper waters during the colder months (Bigelow and Schroeder, 1953).

Sculpin eggs adhere to each other in irregular masses and sink to the bottom. Eggs may be found in a range of habitats and depths (Bigelow and Schroeder, 1953).

Silver hake (*Merluccius bilinearis*)

Source: NASA, 2001

The silver hake is a member of the Gadidae family. The silver hake is a demersal species that is often found in dense schools from Nova Scotia to North Carolina (Morse et al., 1999). They are voracious predators and have a wide range that depends on the abundance of prey (Bigelow and Schroeder, 1953). They have been found at depths ranging from the tideline to over 700 m (Scott and Crossman, 1973).

Silver hake spawn in open water in a wide range of depths and temperatures. Eggs and larvae are pelagic and drift with the currents. Juveniles and adults are primarily demersal. Silver hake migrate closer to shore in the spring and summer, and overwinter in deeper waters (Morse et al., 1999).

Striped bass (*Morone saxatilis*)



Source: Froese and Pauly, 2000

The striped bass is a member of the temperate bass family, Moronidae. Both migratory and nonmigratory populations span the western Atlantic coast, ranging from the St. Lawrence River, Canada, to the St. John's River in Florida (Scott and Scott, 1988). The striped bass has long been an important commercial and recreational species. The perceived decline in striped bass populations was the reason behind the creation of the Atlantic

States Marine Fisheries Commission in 1942 (Miller, 1995).

Striped bass are common along mid-Atlantic coastal waters. They are anadromous fish that spend most of the year in saltwater but use the upper fresh and brackish water reaches of estuaries as spawning and nursery areas in spring and summer (Setzler et al., 1980). The principal spawning areas for striped bass along the Atlantic coast are the major tributaries of the Chesapeake Bay and the Delaware and Hudson rivers (Shepherd, 2000). The timing of spawning may be triggered by an increase in water temperature, and generally occurs from April to June (Fay et al., 1983c). Spawning behavior consists of a female surrounded by up to 50 males at or near the surface (Setzler et al., 1980). Eggs are broadcast loosely in the water and fertilized by the males. Females may release an estimated 15,000 to 40.5 million eggs, depending on the size of the female (Mansueti and Hollis, 1963; Jackson and Tiller, 1952).

Striped bass eggs are semibuoyant, and require minimum water velocities to remain buoyant. Eggs that settle to the bottom may become smothered by sediment (Hill et al., 1989). Depending on water temperature, fertilized eggs hatch anywhere from 29 to 80 hours after fertilization (Hardy, 1978). The duration of larval development is influenced by temperature; water temperatures ranging from 24 to 15°C correspond to larval durations of 23 to 68 days (Rogers et al., 1977). One study in Setzler et al. (1980) reported a 6% probability of survival for egg and yolk-sac stages of development, and a 4% probability of survival for the post yolk-sac stage.

At 30 mm, most striped bass enter the juvenile stage. Juveniles begin schooling in larger groups after 2 years of age (Bigelow and Schroeder, 1953). Migratory patterns of juveniles vary with locality (Setzler et al., 1980). In the Delaware and the Hudson rivers, young-of-year migrate downstream from their spawning grounds to the tidal portions of the rivers to spend their first summer (Able and Fahay, 1998). In the Delaware River, young-of-year may spend 2 or more years within the estuary before joining the offshore migratory population (Miller, 1995). Similar trends were found in the Hudson River, where individuals stayed up to 3 years in estuaries before migrating offshore (Able and Fahay, 1998).

Striped killifish (*Fundulus majalis*)

Source: Maryland DNR, 2001

The striped killifish belongs to the Fundulidae (mummichog and killifish) family and is a small (<15 cm) fish that inhabits bays, estuaries, coastal marshes, and tidal creeks (Froese and Pauly, 2000). Killifish are common along the Atlantic coast from Massachusetts to North Carolina and are most often seen in shallow water over sandy substrate (Abraham, 1985). Striped killifish feed on the marsh surface during high tide (Abraham, 1985).

Striped killifish spawn in still, shallow water close to shore and in ponds. The female actively buries her eggs (Abraham, 1985).

Tautog (*Tautoga onitis*)

Source: NOAA, 2001a

The tautog is a member of the Labridae family and is found in coastal areas from New Brunswick south to South Carolina. It is most abundant from Cape Cod, Massachusetts, to the Delaware Estuary (ASMFC, 2000). Tautog are most frequently found close to shore, preferring rocky areas or other discontinuities such as pilings, jetties, or wrecks and salinities of greater than 25 ppt.

Tautog migrate inshore in the spring to spawn in inshore waters. Spawning generally occurs between mid-May and August, peaks in June (Auster, 1989), and primarily takes place at the mouths of estuaries and along the inner continental shelf (Able and Fahay, 1998; Steimle and Shaheen, 1999). The eggs are buoyant, and hatch out in approximately 2 to 3 days (Auster, 1989).

Larvae migrate vertically in the water column, surfacing during the day and remaining near the bottom at night. As they get older, they become more benthic (Steimle and Shaheen, 1999). Small juveniles will remain in estuaries year-round, becoming torpid over the winter (Jury et al., 1994), while larger ones will join adults in deeper water. Small juveniles prefer vegetated habitats in depths of less than 1 m. Older juveniles and adults inhabit reef-like habitats that provide some type of cover (Steimle and Shaheen, 1999).

Tautog do not tend to migrate far offshore; however, adults move to deeper water in the fall, responding to decreases in water temperature. Adults return to coastal waters and estuaries to spawn when waters warm in the spring (Steimle and Shaheen, 1999).

Threespine stickleback (*Gasterosteus aculeatus*)



Source: Royal BC
Museum, 2001

The threespine stickleback belongs to the Gasterosteidae (sticklebacks) family and is a resident of coastal and estuarine waters, although it can be found in open water and freshwater. Its preferred habitats are tidal marshes and creeks, brackish pools and lagoons, and weedy, shallow shores. It is a pelagic species commonly associated with submerged aquatic vegetation such as eelgrass and rockweed (Bigelow and Schroeder, 1953).

Threespine stickleback is considered an anadromous species; it migrates into estuaries or freshwater to spawn in March and April (Able and Fahay, 1998). Males build nests in sheltered shoals and the eggs stick to the nests and each other. The male guards the nests until the fry are able to swim (Bigelow and Schroeder, 1953). Larvae then disperse into shallow water with dense vegetation (Wang, 1986). Adults that survive return to more saline waters after spawning, although they have a high rate of mortality after spawning (Able and Fahay, 1998).

White perch (*Morone americana*)



Source: New York Sportfishing and
Aquatic Resources Educational
Program, 2001

The white perch belongs to the Moronidae family. The geographic range of the white perch extends from the upper St. Lawrence and Great Lakes to South Carolina (Scott and Scott, 1988; Able and Fahay, 1998). Adults can be found in a wide range of habitats, but they exhibit a preference for shallow water during warmer months (Stanley and Danie, 1983). In the winter months, adults can be found in deeper, saline waters (Beck, 1995).

White perch are semianadromous, overwintering in deeper estuarine waters and migrating seasonally in the spring to spawn. Spawning occurs from April through early June in shallow waters of upstream brackish and freshwater tributaries (Scott and Crossman, 1973).

Larvae are pelagic, remaining slightly below the surface of the water. Juveniles become increasingly demersal with size (Wang and Kernehan, 1979) and school in shallow, inshore waters through the summer. During the fall, juveniles tend to move offshore into more brackish, deeper waters to overwinter.

At the larval stage, white perch feed mainly on plankton. Adults feed on a variety of prey, including shrimp, fish, and crab. Their diet composition changes with seasonal and spatial food availability (Beck, 1995).

Unlike most other species, white perch did not suffer a drastic population decline in the past century. Because of their abundance, white perch are valuable for commercial fisheries and the recreational fishing industry. Their heartiness and abundance is due to their proliferation, early maturation, ability to utilize a large spawning and nursery ground, and tolerance of poor water quality (Beck, 1995).

Windowpane (*Scophthalmus aquosus*)



Source: NOAA, 2001b

The windowpane belongs to the Scophthalmidae (one of the flounder families) family and is found from the Gulf of St. Lawrence to Florida, inhabiting estuarine and shallow continental shelf waters less than 56 m deep (Able and Fahay, 1998). They have been found in areas with sandy bottoms, water temperatures ranging from 3 to 21°C, and salinities of 27 to 31 ppt (Kaiser and Neuman, 1995).

Spawning occurs over the continental shelf and in estuaries, and windowpane will not spawn in waters over 20°C (Kaiser and Neuman, 1995). The timing of spawning varies with location; in mid-Atlantic Bight waters, spawning occurs from April through December, peaking in May and October, while on Georges Bank spawning occurs during summer and peaks in July and August (Hendrickson, 2000). Eggs are buoyant and hatch out in 8 days at a water temperature of 11°C (Chang et al., 1999). Eggs and larvae are planktonic, but movements are poorly understood. Juveniles appear to use estuaries as nursing areas, and then return to offshore waters in the fall (Kaiser and Neuman, 1995).

Researchers disagree on how extensively windowpane migrate. Although they have been found to travel 130 km in a few months, most researchers agree that windowpane generally do not migrate long distances. Juveniles along Georges Bank exhibit seasonal migration to deeper waters in late autumn to overwinter (Chang et al., 1999).

Winter flounder (*Pleuronectes americanus*)



Source: Maine Dept. of Marine Resources, 2001

The winter flounder is a benthic flatfish of the Pseudopleuronectes (one of the flounder families) family and is found in estuarine and continental shelf habitats. Its range extends from the southern edge of the Grand Banks south to Georgia (Buckley, 1989b). It is a bottom feeder, occupying sandy or muddy habitats and feeding on bottom-dwelling organisms (Froese and Pauly, 2000).

The winter flounder is essentially nonmigratory, but there are seasonal patterns in movements within the estuary. Winter flounder south of Cape Cod generally move to deeper, cooler water in summer and return to shallower areas in the fall, possibly in response to temperature changes (Howe and Coates, 1975; Scott and Scott, 1988).

Spawning occurs between January and May in New England, but peaks in Massachusetts in February and March (Bigelow and Schroeder, 1953). Spawning habitat is generally in shallow water over a sandy or muddy bottom (Scott and Scott, 1988). Adult fish tend to leave the shallow water in autumn to spawn at the head of estuaries in late winter. The majority of spawning takes place in a salinity range of 31 to 33 ppt and a water temperature range of 0 to 3°C. Females will usually produce between 500,000 and 1.5 million eggs annually, which sink to the bottom in clusters (Bigelow and Schroeder, 1953).

Larvae depend on light and vision to feed during the day and do not feed at night (Buckley, 1989b). Juveniles tend to remain in shallow spawning waters, and stay on the ocean bottom (Scott and Scott, 1988).

Food sources and predator-prey requirements

Along with physical habitat needs, the fishes discussed above also need a plentiful food source to sustain them. Predator-prey relationships within an ecosystem drive the flow of nutrients and carbon and must be balanced to be sustainable. A brief summary of the predator-prey relationships for species that experience I&E losses is presented in Table 4-5.

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility.

Species	Prey	Predators	References
Alewife	Small fish, zooplankton, fish eggs, amphipods, mysids. Juveniles feed mainly on plankton.	Many fish, including striped bass, weakfish and rainbow trout	Waterfield, 1995
American plaice	Sea urchins, sand dollars, and brittle stars; young feed on plankton, diatoms, and copepods.		Bigelow and Schroeder, 1953, Johnson et al., 1999a
American sand lance	Mainly copepods.	Dolphinfish, Greenland cod, silver hake, white hake, Atlantic cod, yellowtail flounder, and longhorn sculpin, whales, and porpoises	Froese and Pauly, 2000; Auster and Stewart, 1986
Atlantic cod	Fry eat copepods, amphipods, larvae, and small crustaceans; juveniles eat larger crustaceans; and adults over 50 cm eat fish, including smaller cod, as well as invertebrates.	Larger cod, squid, pollock	Grant and Brown, 1998; Scott and Scott, 1988
Atlantic herring	Small planktonic copepods in the first year then copepods, fish eggs, pteropods (small molluscs), and the larvae of mollusks and fish.	Almost all pelagic predators, as well as many seabirds, marine mammals, and bottom dwellers (eggs only)	Scott and Scott, 1988
Atlantic mackerel	Zooplankton, shrimp, crab larvae, small squid, fish eggs, and young fish such as capelin and herring.	Whales, porpoises, mackerel sharks, threshers, dogfish, tuna, bonito, bluefish, striped bass, and cod	Bigelow and Schroeder, 1953

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Atlantic menhaden	Plankton (primarily diatoms and dinoflagellates).	Almost all piscivorous, recreationally important fish, including cod, pollock, hakes, bluefish, tuna, and swordfish, as well as dolphins, sharks, whales, and birds	Hall, 1995; Scott & Scott, 1988
Atlantic silverside	Copepods, mysids, amphipods, cladocerans, fish eggs, squid, worms, molluscs, insects, algae, and detritus.	Valuable fishery species such as striped bass, bluefish, weakfish, and Atlantic mackerel	Fay et al., 1983a; McBride, 1995
Bay anchovy	Copepods and other zooplankton, as well as small fishes and gastropods.	Striped bass, weakfish, jellyfish, birds	Morton, 1989
Blue mussel	Phytoplankton.	Birds such as diving ducks, gulls, American oystercatcher; aquatic predators such as American lobster, crabs, starfish, whelks, tautog, cunner, and other species of fish	Newell, 1989
Blueback herring	Shrimp, zooplankton, finfish.	Many estuarine species including striped bass, weakfish, bluefish	Fay et al., 1983b
Bluefish	Juveniles: shrimp, anchovies, killifish, and silversides; Adults: squid, shrimp, crabs, shad, herrings, Atlantic menhaden, silver hake, spot, butterfish, smaller bluefish, as well as many other fish species.	Larger bluefish, sharks, tuna, and swordfish	Pottern et al., 1989
Butterfish	Small fishes, squids, coelenterates, and ctenophores	Haddock, silver hake, bluefish, swordfish, weakfish, sharks, skates, and long-finned squid.	Cross et al., 1999
Cunner	Mussels, small lobsters, and sea urchins in addition to plant material.	Other shore fish such as sculpins, seabirds	Maine Division of Marine Resources, 2001; Lawton, 2000
Fourbeard rockling	Flatfishes, amphipods, decapods, copepods, mysids, shrimps, isopods and other small crustaceans.	Cod, mackerel, and other predatory fish	Froese and Pauly, 2000; Bigelow and Schroeder, 1953
Fourspot flounder	Small fish, squid, shrimp, crabs, shellfish, and worms.		Bigelow and Schroeder, 1953

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Grubby	Omnivorous, eating annelid worms, shrimp, crabs, copepods, snails, molluscs, ascidians, and small fish such as alewives, cunners, eels, mummichogs, lance, silversides, sticklebacks, and tomcod.	Predatory fish	Bigelow and Schroeder, 1953
Hogchoker	Annelid worms and crustaceans.		Bigelow and Schroeder, 1953
Longhorn sculpin	Shrimp, crab, amphipods, hydroids, annelid worms, mussels, squid, ascidians, and fish such as alewives, cunners, eels, mummichogs, herring, mackerel, menhaden, puffers, lance, scup, and others.	Atlantic cod	Froese and Pauly, 2000; Bigelow and Schroeder, 1953
Lumpfish	Ctenophores, medusae, small crustaceans, polychaetes, jelly fish and small fishes.	Seals	Bigelow and Schroeder, 1953; Froese and Pauly, 2000
Northern pipefish	Copepods, amphipods, fish eggs, small fry.		Bigelow and Schroeder, 1953
Northern searobin	Shrimps, crabs, other crustaceans, squid, bivalves and small fishes.	Sand devils	Froese and Pauly, 2000
Pollock	Juveniles: primarily crustaceans, also small fish and mollusks; Adults: euphausiids, fish (especially Atlantic herring) and mollusks.		Cargnelli et al., 1999
Radiated shanny	Juveniles: copepods; Adults: mostly nereids, also capelin eggs, scaleworms, and amphipods.	Atlantic cod; Juveniles: grubby	Scott and Scott, 1988
Rainbow smelt	Larvae and Juveniles: Copepods, planktonic crustaceans; Adults: small mummichogs, cunner, anchovies, sticklebacks, silversides, and alewives, as well as euphausiids, amphipods, and polychaetes.	Striped bass, bluefish. Eggs are eaten by mummichog and fourspine stickleback	Buckley, 1989a
Red hake	Shrimps, amphipods and other crustaceans, also on squid and herring, flatfish, mackerel and others.	Striped bass, spiny dogfish, goosefish, white hake, silver hake, sea raven, harbor porpoise, and other predators	Steimle et al., 1999; Froese and Pauly, 2000

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Rock gunnel	Small crustaceans, polychaetes, molluscs and fish eggs.	Cod, pollock	Bigelow and Schroeder, 1953; Froese and Pauly, 2000
Shorthorn sculpin	Crab and other crustaceans, shrimp, sea urchins, worms, and fry of other fish.	Atlantic cod	Froese and Pauly, 2000; Bigelow and Schroeder, 1953
Silver hake	Fish (alewife, butterfish, cunner, herring, mackerel, menhaden, scup, silversides, smelt, young of its own species), crustaceans, shrimp.	Bluefish, butterfish	Bigelow and Schroeder, 1953; Morse et al., 1999
Striped bass	Mysid shrimp and smaller fish species such as herring, silversides, and anchovies; Larvae feed primarily on copepods.	Sea lamprey, striped bass, silver hake, bluefish, copepods	Miller, 1995
Striped killifish	Crustaceans and polychaetes.	Wading birds, aerial searching birds, piscivorous ducks, crabs, and many predatory fishes. Fishes include white perch, summer flounder, striped bass, bluefish, and red drum. Birds include herons, egrets, terns, gulls, and least common terns	Abraham, 1985
Summer flounder	Small fish, small shelled mollusks, worms, sand dollars, squids, crabs, shrimp, and other crustaceans.	Larvae and juveniles: spiny dogfish, cod, goosefish, hake, sea raven, longhorn sculpin, and fourspot flounder	Bigelow and Schroeder, 1953
Tautog	Mussels, small crustaceans and other molluscs. Juveniles feed on amphipods and copepods.	Smooth dogfish, barndoor skate, red hake, sea raven, goosefish, and seabirds	Jury et al., 1994; Steimle and Shaheen, 1999
Threespine stickleback	Omnivorous. Small invertebrates, fish fry, fish eggs, shrimp, small squids, and diatoms.	Sea trout, whiting, eels	Bigelow and Schroeder, 1953; Froese and Pauly, 2000
White perch	Variety of prey, including shrimp, fish, and crab. Their diet composition changes with seasonal and spatial food availability. Larvae feed mainly on plankton.	Striped bass, bluefish, weakfish, walleye, copepods	Beck, 1995

Table 4-5. Predator-prey relationships for species commonly impinged or entrained at the Pilgrim facility (cont.).

Species	Prey	Predators	References
Windowpane	Young consume mysids, while adults feed on sand shrimp, small fish (up to 10 cm), crustaceans, molluscs, and seaweed.	Spiny dogfish, thorny skate, goosefish, Atlantic cod, black sea bass, weakfish, and summer flounder	Chang et al., 1999
Winter flounder	Benthic organisms such as shrimp, amphipods, crabs, urchins and snails.	Larger estuarine and coastal fish such as striped bass and bluefish	Buckley, 1989b; Froese & Pauly, 2000
Yellowtail flounder	Small crustaceans (including amphipods, shrimps, and mysids), small shellfish, and worms.	Spiny dogfish, skates, Atlantic halibut, fourspot flounder, goosefish, silver hake, bluefish and sea raven	Bigelow and Schroeder, 1953; Johnson et al., 1999b

4.3 Step 3: Identify Potential Habitat Restoration Alternatives to Offset I&E Losses

Local experts proposed six types of habitat restoration projects that would offset I&E losses at the Pilgrim facility:

- ▶ improve water quality
- ▶ reduce fishing pressures
- ▶ restore tidal wetlands
- ▶ restore submerged aquatic vegetation
- ▶ improve anadromous fish passage
- ▶ create artificial reefs.

Each of these potential restoration projects provide benefits to the aquatic community, and are described below.

Improve water quality

Water quality plays a major role in determining whether fish can survive in a given water body. Water quality can be compromised by high levels of industrial pollutants, nutrients from wastewater treatment plants and failing septic systems, and extreme temperatures. Some examples of water quality improvement projects may include (but are not limited to):

- ▶ remove nitrogen and phosphorus at wastewater treatment plants
- ▶ improve storm water management

- ▶ repair or replace failing septic systems
- ▶ provide better “pump-out” services to recreational and commercial boaters to dispose of their boat waste in a safe and sanitary manner
- ▶ limit discharges of hazardous materials from industrial facilities
- ▶ limit thermal discharges.

Any measures to improve water quality by limiting the amount of pollutants in the estuaries surrounding the Pilgrim facility benefit the aquatic ecosystem. Reducing pollutant levels will increase survival rates for invertebrates, fish, and other animals that depend on the estuarine ecosystem. Improving water quality can restore fish and shellfish habitats that were previously limited or uninhabitable because of toxicity or intolerance to polluted conditions.

Reduce fishing pressures

Fish that support commercial or recreational fisheries are prone to high mortality rates because of fishing pressures. These species can benefit from reduced fishing. Some potential projects that could be implemented to reduce fishing pressures include closing sensitive areas (such as spawning grounds) to fishing during certain times during the year, or decreasing the number of fishing licenses that are issued. Fishing gear could also be changed to limit the number of unwanted fish caught. For example, fishing nets could be altered to reduce the catch of small or undesirable fish that are caught in existing nets.

Restore tidal wetlands

Tidal wetlands (Figure 4-1) are among the most productive ecosystems in the world (Mitsch and Gosselink, 1993; Broome and Craft, 2000). Tidal wetlands provide valuable habitat for many species of invertebrates and forage fish that serve as food for other species in and near the wetland. Tidal wetlands also provide spawning and nursery habitat for many other fish species, including the Atlantic silverside, striped killifish, threespine stickleback, and mummichog. Other migratory species that use tidal wetlands during their lives include the winter flounder, striped bass, Atlantic herring, and white perch (Dionne et al., 1999). Fish species that have been reported in restored salt ponds and tidal creeks include Atlantic menhaden, blueback herring, Atlantic silverside, striped killifish, and mummichog [Roman et al., (submitted to *Restoration Ecology*)]. Restoring tidal flow to areas where such flows have been restricted has also been shown to reduce the presence of *Phragmites australis*, the invasive marsh grass that has choked out native flora and fauna in coastal areas across the New England seaboard (Fell et al., 2000).

Tidal wetlands restoration typically involves returning tidal flow to marshes or ponds that have restrictions of natural tidewater flow by roads, backfilling, dikes, or other barriers. Eliminating these barriers can restore salt marshes (Figure 4-2), salt ponds, and tidal creeks that provide essential habitat for many species of aquatic organisms. For example, where tidal flow is reduced



Figure 4-1. Tidal creek near Little Harbor, Cohasset, Massachusetts.

Source: MAPC, 2001.



Figure 4-2. Salt marsh near Narragansett Bay, Rhode Island.

Source: Save The Bay, 2001.

by undersized culverts, installing correctly sized and positioned culverts can restore tidal range and proper salinity. In other situations, such as where low-lying property adjacent to salt marsh has been developed, restoring full tidal flow may not be possible because of flood concerns (MAPC, 2001). Salt marshes can also be created by flooding areas in which no marsh habitat previously existed (e.g., tidal wetland creation). However, a study by Dionne et al. (1999) showed that while both created and restored tidal wetlands readily provide habitat for a number of fish, restored tidal wetlands provide much larger and more productive areas of habitat per unit cost than created tidal wetlands.

Restore submerged aquatic vegetation

Submerged aquatic vegetation (SAV) provides vital habitat for a number of aquatic organisms. Eelgrass is the dominant species of SAV along the coasts of New England. It is an underwater flowering plant that is found in brackish and near-shore marine waters (Figure 4-3). Eelgrass can form large meadows or small separate beds that range in size from many acres to just 1 m across (Save The Bay, 2001).

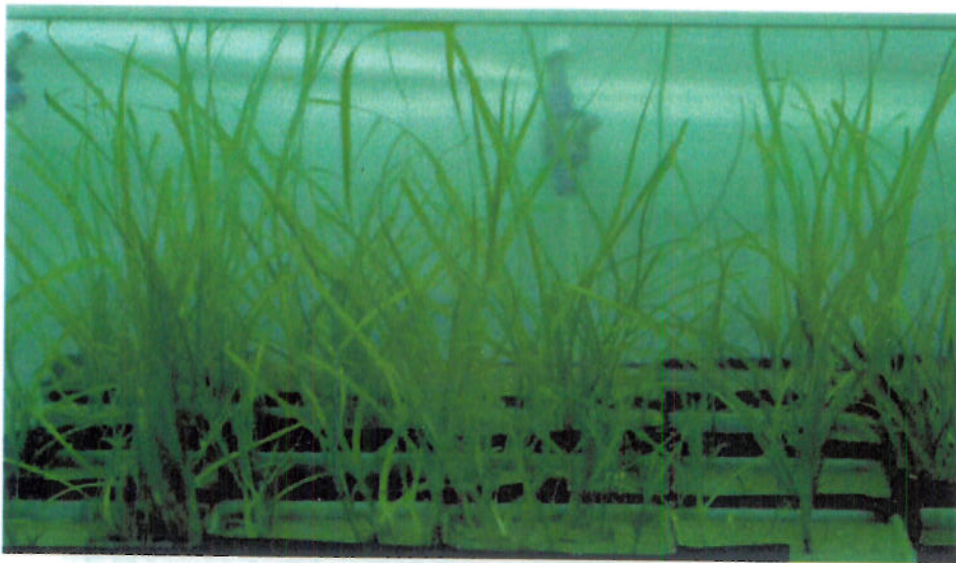


Figure 4-3. Laboratory culture of eelgrass (*Zostera marina*).

Source: Boschker, 2001.

SAV restoration involves transplanting eelgrass shoots and/or seeds into areas that can support their growth. Site selection is based on historical distribution, wave action, light availability, sediment type, and nutrient loading. Improving water quality and clarity, reducing nutrient levels, and restricting dredging may all be necessary to promote sustainable eelgrass beds. Protecting existing SAV beds is a priority in many communities (Save The Bay, 2001).

SAV provides several ecological services to the environment. It has a high rate of leaf growth and provides support for many aquatic organisms as shelter, spawning, and nursery habitat. It is also a food source for herbivorous organisms. The roots of SAV also provide stability to the bottom sediments, thus decreasing erosion and resuspension of sediments into the water column (Thayer et al., 1997). Dense SAV provides shelter for small and juvenile fishes and invertebrates from predators. Small prey can hide deep within the SAV canopy, and some prey species use the SAV as camouflage (Thayer et al., 1997). Species that use SAV beds during early life stages include Atlantic menhaden, striped bass, American eel, tautog, bluefish, summer flounder, weakfish, rainbow smelt, bay scallops, and blue crab (Laney, 1997).

Improve anadromous fish passageways

Anadromous fish spend most of their lives in brackish or saltwater but migrate into freshwater rivers and streams to spawn. Many of the rivers and streams that historically supported anadromous fish spawning have been dammed and are currently inaccessible to migrating fish. Anadromous fish that would benefit from improved access to upstream spawning habitat include alewife, Atlantic salmon (*Salmo salar*), rainbow smelt, sturgeon, white perch, American eel, and American shad.

Improving anadromous fish passage involves many important steps. Dams and barriers connecting estuaries with upstream spawning habitat can be removed or fitted with fish ladders (Figure 4-4). Removing the dam is often preferable because some species, such as rainbow smelt, use fish ladders ineffectively. However, dam removal may not be possible in highly developed areas needing flood control. In addition, restoring stream habitats such as forested riverbank wetlands and improving water quality may also be necessary to restore upstream spawning habitats for anadromous fish (Save The Bay, 2001).

Create artificial reefs

Several species of fish found near the Pilgrim facility use rocky or reef-like habitats with interstices that provide refuge from predators. These habitats can be created artificially with cobbles, concrete, and other suitable materials.

Species that commonly use reef structures for refuge include tautog, cunner, scup, black sea bass, lobsters, and blue mussels (Foster et al., 1994; Castro et al., in press). Both cunner and tautog become torpid at night and require places to hide from their prey. Blue mussels use rocky reefs for attachment.

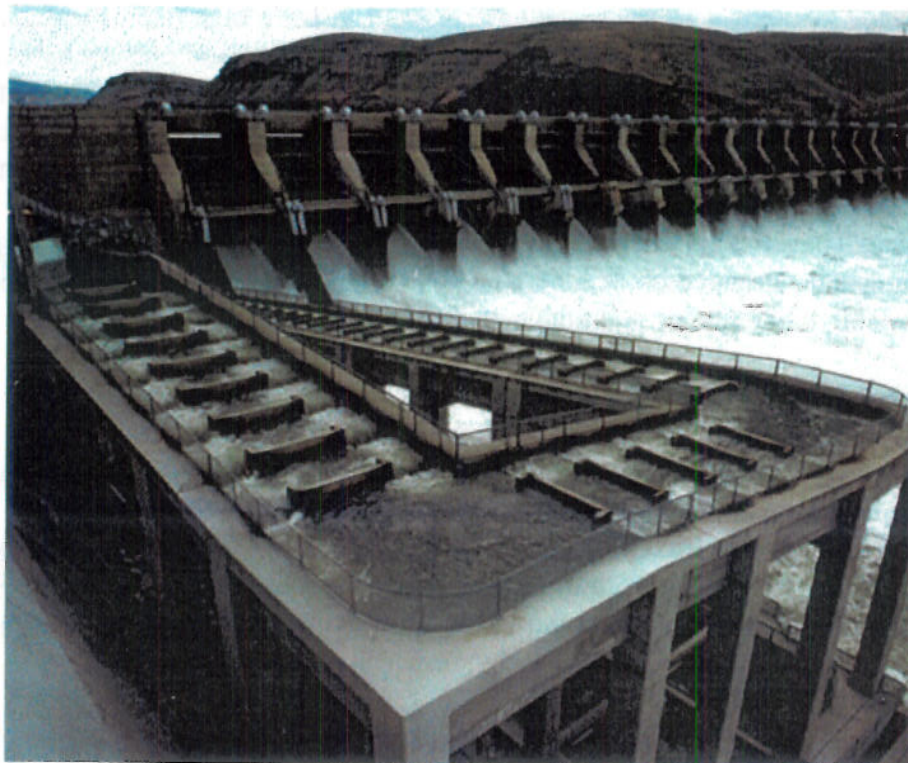


Figure 4-4. Example of a fish ladder at a hydroelectric dam.

Source: Pollock, 2001.

4.4 Step 4: Consolidate, Categorize, and Prioritize Identified Habitat Restoration Alternatives

Habitat restoration alternatives were categorized and prioritized in collaboration with local experts. Meetings were designed to identify the restoration program for each of the major species that are impinged or entrained as a result of cooling water intakes. Meetings were arranged and moderated by Stratus Consulting, and attended by several federal, state, and local organizations (Table 4-6).

Habitat needs and restoration options for each species with significant I&E losses at the facility were discussed. These restoration options were then prioritized for each species by determining what single restoration option would most benefit that species. The higher ranked restoration alternatives for each species are shown in Table 4-7.

Table 4-6. Attendees at the Pilgrim Meeting, September 12, 2001, in Lakeville, Massachusetts.

Attendee	Organization
David Allen	Stratus Consulting
David Mills	Stratus Consulting
Michelle Barron	Stratus Consulting
Bob Green	Massachusetts DEP
Robert Lawton	Massachusetts Division of Marine Fisheries
George Zoto	Massachusetts Watershed Initiative - South Coastal Watersheds
Kathi Rodrigues	National Marine Fisheries Service - Restoration Center
David Webster	U.S. EPA Region I
Sharon Zaya	U.S. EPA Region I
Nick Prodan	U.S. EPA Region I
John Nagle	U.S. EPA Region I

Table 4-7. Restoration alternatives for each Pilgrim species ranked highest by local experts.

Species	Prioritized restoration alternatives
Alewife	Anadromous fish passage
Atlantic herring	Anadromous fish passage
Blueback herring	Anadromous fish passage
Rainbow smelt	Anadromous fish passage (remove dams)
White perch	Anadromous fish passage
Cunner	Artificial reefs, SAV restoration
Sculpin spp.	Artificial reefs, SAV restoration (improve habitat for prey)
Tautog	Artificial reefs, SAV restoration
American sand lance	Tidal wetlands restoration
Atlantic silverside	Tidal wetlands restoration
Bluefish	Tidal wetlands restoration (improve habitat for prey)
Grubby	Tidal wetlands restoration
Striped bass	Tidal wetlands restoration (improve habitat for prey)
Windowpane ^a	Tidal wetlands restoration (improve habitat for prey)
Winter flounder	Tidal wetlands restoration
Threespine stickleback	SAV restoration, tidal wetland restoration
Atlantic mackerel	Reduce fishing pressure, improve water quality
Atlantic menhaden	Reduce fishing pressure, improve water quality
Bay anchovy	Reduce fishing pressure, improve water quality
Butterfish	Reduce fishing pressure, improve water quality
All species	Improve water quality

a. Improved water quality later became the chosen restoration alternative for windowpane because they inhabit depths greater than accessible to tidal wetland restoration.

Table 4-12. Average abundance from Rhode Island SAV sites for Pilgrim species that would benefit most from SAV restoration.

Species	Species abundance (# fish per 100 m ² of SAV habitat) ^a	
	Low quality SAV habitats	High quality SAV habitats
Atlantic tomcod	0.52	1.77
Pollock	no obs.	no obs.
Northern pipefish	0.23	3.03
Threespine stickleback	no obs.	19.67

a. High quality habitats are defined as areas with eelgrass shoot densities > 100 per m² and shoot biomass (wet) > 100 g/m². Low quality habitats do not meet these criteria.

Source: personal communication, J. Hughes, NOAA, Marine Biological Laboratory, 2001.

Heck et al., 1989 — Species abundance in Nauset Marsh (Massachusetts) estuarine complex SAV

Heck et al. (1989) provide capture totals for day and night trawl samples taken between August 1985 and October 1986 in the Nauset Marsh Estuarine Complex in Orleans/Eastham, Massachusetts, including two eelgrass beds: Fort Hill and Nauset Harbor. As in the other SAV sampling efforts, an otter trawl was used for the sampling, but with slightly larger mesh size openings in the cod end liner (6.3 mm versus 3.0 mm) than in Hughes et al. (2001) or Wyda et al. (in press).

With the reported information on the average speed, duration, and number of trawls used in each sampling period and an estimate of the width of the SAV habitat covered by the trawl from one of the study authors (personal communication, M. Fahay, NOAA, 2001), abundance estimates per 100 m² of SAV habitat were calculated.

Heck et al. (1989) also report that the dry weight of the SAV shoots is over 180 g/m² at both the Fort Hill and Nauset Harbor eelgrass habitat sites. Therefore, these locations would fall into the high SAV habitat category used in Wyda et al. (in press) and Hughes et al. (2000) because the dry weight exceeds the wet weight criterion of 100 g/m² used in those studies.

Finally, Heck et al. (1989) provide separate monthly capture results from their trawls. The maximum monthly capture results for each species was used for the abundance estimates from this sampling. Because these maximum values generally occur in the late summer months, sampling time is consistent with the results from Wyda et al. (in press) and Hughes et al. (2000).

The species abundance values estimated from the sampling of the Fort Hill and Nauset Harbor SAV habitats are presented in Table 4-13.

Table 4-13. Average abundance in Nauset Marsh Estuarine Complex SAV for Pilgrim species that would benefit most from SAV restoration.

Species	Species abundance (# fish per 100 m ²) ^a	
	Fort Hill — High quality SAV	Nauset Harbor — High quality SAV
Atlantic tomcod	no obs.	0.08
Pollock	no obs.	no obs.
Northern pipefish	0.68	6.11
Threespine stickleback	5.92	47.08

a. High quality habitats are defined as areas with eelgrass shoot densities > 100 per m² and shoot biomass (wet) > 100 g/m².

Source: Heck et al., 1989.

4.5.1.2 Adjusting SAV sampling results to estimate annual average increase in production of age-1 fish

Sampling-based abundance estimates were adjusted to account for:

- ▶ sampling efficiency
- ▶ capture of life stages other than age 1
- ▶ differences in the productivity of restored versus natural SAV habitat.

The basis and magnitude of the adjustments are discussed in the following sections.

Adjusting for sampling efficiency

Fish sampling techniques are unlikely to capture and/or record all of the fish present in a sampled area because some fish avoid the sampling gear and some are captured but not collected and counted. The sampling efficiency for otter trawls is approximately 40% to 60% (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001). A conservative sampling efficiency of 40% was assumed for this HRC analysis. Therefore, the SAV sampling abundance estimates were multiplied by 2.5 (i.e., divided by 40%). This assumption increases SAV productivity estimates and lowers SAV restoration cost estimates.

Adjusting sample abundance estimates to age-1 life stages

All sampled life stages were converted to age-1 equivalents for comparison to I&E losses, which were expressed as age-1 equivalents. The average life stage of the fish caught in the Buzzards Bay (Wyda et al., in press) and Rhode Island coastal salt pond (Hughes et al., 2000) was juveniles (i.e., life stage younger than age 1) (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001). Since the same sampling technique and gear was used in Heck et al. (1989), juveniles were assumed to be the average life stage captured in this study as well.

Table 4-15. Final estimates of the increase in production of age-1 fish for Pilgrim species that would benefit most from SAV restoration (cont.).

Species	Source of initial species abundance estimate	Species abundance estimate per 100 m ² of SAV	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Expected increase in production of age-1 fish per 100 m ² of restored SAV
Northern pipefish	Hughes et al. (2000) — RI coastal ponds (high SAV)	3.03	2.5	0.5352	1.0	4.06
	Wyda et al. (in press) — Buzzards Bay (low SAV)	0.19	2.5	0.5352	1.0	0.25
	Wyda et al. (in press) — Buzzards Bay (high SAV)	0.99	2.5	0.5352	1.0	1.32
	Species average					2.50
Threespine stickleback	Heck et al. (1989) — Fort Hill	5.92	2.5	0.5284	1.0	7.82
	Heck et al. (1989) — Nauset Harbor	47.08	2.5	0.5284	1.0	62.19
	Hughes et al. (2000) — RI coastal ponds (high SAV)	19.67	2.5	0.5284	1.0	25.98
	Wyda et al. (in press) — Buzzards Bay (low SAV)	0.22	2.5	0.5284	1.0	0.29
	Wyda et al. (in press) — Buzzards Bay (high SAV)	0.13	2.5	0.5284	1.0	0.17
	Species average					19.29
Pollock	no obs.					

4.5.2 Estimates of Increased Age-1 Fish Production from Tidal Wetland Restoration

Tidal wetlands provide a diversity of habitats such as open water, subtidal pools, ponds, intertidal waterways, and tidally flooded meadows of salt tolerant species such as *Spartina alterniflora* and *S. patens*. These habitats provide forage, spawning, nursery, and refuge for a large number of fish species. Table 4-16 identifies the I&E losses for fish species at Pilgrim that would benefit most from tidal wetland restoration, along with average I&E losses for the period 1974-1999, arranged by number of fish lost.

Table 4-16. Pilgrim species that would benefit most from tidal wetland restoration.

Species	Annual average I&E loss of age 1 equivalents (1974-1999)	Percentage of annual average I&E loss across all fish species
American sand lance	4,116,285	28.55%
Winter flounder	210,715	1.46%
Atlantic silverside	25,929	0.18%
Grubby	879	0.01%
Striped killifish	90	0.00%
Striped bass	9	0.00%
Bluefish	2	0.00%
Total	4,353,909	30.20%

Restricted tidal flows increase the dominance of *Phragmites australis* by reducing tidal flushing and lowering salinity levels (Buzzards Bay Project National Estuary Program, 2001). *Phragmites* dominance restricts fish access to and movement through the water, decreasing overall productivity of the habitat. Therefore, for the purpose of this HRC valuation, tidal wetland restoration focuses on returning natural tidal flows to currently restricted areas. Examples of actions that can restore tidal flows to currently restricted tidal wetlands include the following:

- ▶ breaching dikes created to support salt hay farming or to control mosquitos
- ▶ installing properly sized culverts in areas currently lacking tidal exchange
- ▶ removing tide gates on existing culverts
- ▶ excavating dredge spoil covering former tidal wetlands.

No identified studies quantified increased production following implementation of these types of restoration actions for tidal wetlands. Therefore, fish abundance estimates taken from studies of tidal wetlands were used to estimate the fish increase in production that can be gained through restoration. The following subsections present the sampling data and subsequent adjustments made to calculate the expected increased in age-1 production of fish species.

4.5.2.1 Fish species abundance estimates in tidal wetlands habitats

Results from tidal wetland sampling efforts in Rhode Island were used to calculate increased production. Available sampling results from Connecticut (Warren et al., submitted to *Restoration Ecology*) and New Hampshire and Maine coasts (Dionne et al., 1999) were not used. The Connecticut results were omitted because time constraints prevented the conversion of capture results into abundance estimates per unit of tidal wetland area. The New Hampshire and Maine results were omitted because the study locations were too distant from the Pilgrim facility.

Roman et al. (submitted to *Restoration Ecology*) — Species abundance at Sachuest Point tidal wetland, Middletown, Rhode Island

Roman et al. (submitted to *Restoration Ecology*) sampled the fish populations in a 6.3 ha unrestricted tidal wetland at Sachuest Point in Middletown, Rhode Island. The sampling was conducted during August, September, and October of 1997, 1998, and 1999 using a 1 m² throw trap in the creeks and pools of each area during low tide after the wetland surface had drained. Additional sampling was conducted monthly in both the unrestricted and restricted parcels from June through October in 1998 and 1999 using 6 m² bottomless lift nets to sample the flooded wetland surface. The report presents the results of this sampling as abundance estimates of each fish species per square meter (Table 4-17).

Table 4-17. Abundance estimates from the unrestricted tidal wetlands at Sachuest for Pilgrim species that would benefit most from tidal wetlands restoration.

Species	Sampling technique	Fish density estimates in unrestricted tidal wetlands (fish per m ²)		
		1997	1998	1999
American sand lance	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Winter flounder	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Atlantic silverside	throw trap	1.23	0.20	0.07
	lift net	no sampling	no obs.	no obs.
Grubby	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Striped killifish	throw trap	0.70	0.17	0.55
	lift net	no sampling	0.01	0.01
Striped bass	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.
Bluefish	throw trap	no obs.	no obs.	no obs.
	lift net	no sampling	no obs.	no obs.

Source: Roman et al. (submitted to *Restoration Ecology*).

Roman et al. (submitted to *Restoration Ecology*) also sampled a smaller portion of the wetland where tidal flows had recently been restored. However, these results were not used because the sampling most likely was conducted prior to the system reaching full productivity.

Raposa (in press) — Galilee Marsh, Naragansett Rhode, Island

Raposa (in press) sampled the fish populations in the Galilee tidal wetland monthly from June through September of 1997, 1998, and 1999 using 1 m² throw trap in the creeks and pools in the tidal wetland parcels during low tide after the wetland surface had drained. Raposa presents the sampling results as fish species abundance expressed as number of fish per square meter. As with the results from Roman et al. (submitted to *Restoration Ecology*), results from a recently restored portion of the wetland were not used in this HRC to avoid a downward bias in the species density results. The results from this sampling effort are presented in Table 4-18 for the Pilgrim species that would benefit most from tidal wetlands restoration.

Table 4-18. Abundance estimates from the unrestricted tidal wetlands at Galilee for Pilgrim species that would benefit most from tidal wetland restoration.

Species	Sampling technique	Fish density estimates in unrestricted tidal wetlands (fish per m ²)		
		1997	1998	1999
American sand lance	throw trap	no obs.	no obs.	no obs.
Winter flounder	throw trap	no obs.	no obs.	no obs.
Atlantic silverside	throw trap	4.78	1.73	14.38
Grubby	throw trap	no obs.	no obs.	no obs.
Striped killifish	throw trap	4.35	3.50	12.40
Striped bass	throw trap	no obs.	no obs.	no obs.
Bluefish	throw trap	no obs.	no obs.	no obs.

Source: Raposa, in press.

K. Raposa, Naragansett Estuarine Research Reserve, personal communication, 2001 — Coggeshall Marsh, Prudence Island, Rhode Island

Discussions with Kenny Raposa of the Naragansett Estuarine Research Reserve (NERR) revealed that additional fish abundance estimates from tidal wetland sampling were available for the Coggeshall Marsh located on Prudence Island in the NERR. These abundance estimates were based on sampling conducted in July and September 2000. The sampling of the Coggeshall tidal wetland was conducted using 1 m² throw traps in the tidal creeks and pools of the wetland during ebb tide after the wetland surface had drained (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001). The sampling results from this effort are presented in Table 4-19 for the Pilgrim species that would benefit most from tidal wetlands restoration.

The sampling efficiencies of bottomless lift nets for individual fish species are provided in Rozas (1992), and are 93% for striped mullet (*Mugil cephalus*), 81% for gulf killifish (*Fundulus grandis*), and 58% for sheepshead minnow (*Cyprinodon variegatus*). The average of these three sampling efficiencies is 77%, which corresponds to a sampling efficiency adjustment factor of 1.3 (i.e., $1.0/0.77$).

Lastly, although specific studies of the sample efficiency of a beach seine net were not identified, an estimated range of 50% to 75% was provided by the staff involved with the Rhode Island coastal pond survey (personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2002). Using the lower end of this range as a conservative assumption, a sample efficiency adjustment factor of 2.0 (i.e., $1.0/0.5$) was applied for the abundance estimates for both the Rhode Island juvenile finfish survey and the Rhode Island coastal pond survey.

Conversion to age-1 life stage

The sampling techniques described in Section 4.5.2.1 are intended to capture juvenile fish (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001). That juvenile fish were the dominant age class taken was confirmed by the researchers involved in these efforts (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001; personal communication, C. Powell, Rhode Island Department of Environmental Management, 2001; and personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2001). As a result, the sampling results presented in Section 4.5.2.1 required adjustment to account for expected mortality between the juvenile and age-1 life stages. The information used to develop these survival rates and the final life stage adjustment factors are presented in Table 4-22.

Table 4-22. Life stage adjustment factors for Pilgrim species — Tidal wetland restoration.

Species	Oldest life stage before age 1 in I&E model	Estimated survival rate to age 1	Life stage captured in tidal wetland sampling efforts	Estimated life stage adjustment factor
American sand lance	larvae	0.0298	juvenile	0.5149
Winter flounder	juvenile	0.2903	juvenile	0.2903
Atlantic silverside	larvae	0.0044	juvenile	0.5022
Grubby	larvae	0.0180	juvenile	0.5090
Striped killifish	larvae	0.0949	juvenile	0.5474
Striped bass ^a	juvenile	0.5361	juvenile	0.5361
Bluefish	juvenile	0.0103	juvenile	0.0103

a. Information in the I&E model is available for two juvenile life stages for striped bass. The data for the older juvenile life stage were used.

Adjusting for differences between restored and undisturbed habitats

Restoring full tidal flows rapidly eliminates differences in fish populations between unrestricted and restored sites (Roman et al., submitted to *Restoration Ecology*), resulting in very similar species composition and density (Dionne et al., 1999; Fell et al., 2000; Warren et al., submitted to *Restoration Ecology*). However, a lag can occur following restoration (Raposa, in press). Therefore, an adjustment factor of 1.0 was used, signifying that no quantitative adjustment was necessary.

Adjusting sampled abundance for timing and location of sampling

At high tide, fish in a tidal wetland have access to the full range of habitats, including the flooded vegetation, ponds, and creeks that discharge into or drain the wetland. In contrast, at low tide, fish are restricted to tidal pools and creeks. Therefore, sampling conducted at low tide represents a larger area of tidal wetlands than the sampled area. Abundance estimates based on samples taken at low tide were therefore divided by the inverse of the proportion of subtidal habitat to total wetland habitat. In contrast, no adjustment was applied to abundance estimates based on samples such as those from lift nets or seines, taken at high tide or in open water offshore. The site-specific adjustment factors in Table 4-23 were based on information regarding the proportion of each tidal wetland that is subtidal habitat (personal communication, K. Raposa, Naragansett Estuarine Research Reserve, 2001).

Table 4-23. Adjustment factors for tidal wetland sampling conducted at low tide.

Tidal wetland	Ratio of open water (creeks, pools) to total habitat in the wetland	Adjustment factor
Sachuest Marsh	0.055	18.2
Galilee Marsh	0.084	11.9
Coggeshall Marsh	0.052	19.2

4.5.2.3 Final estimates of annual average age-1 fish production from tidal wetland restoration

Table 4-24 presents the final estimates of annual increased production of age-1 fish resulting from tidal wetland restoration for Pilgrim species that would benefit most from tidal wetland restoration.

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration.

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
American sand lance	no obs.							
Winter flounder	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall - July 2000	0.10	1.6	0.2903	1	19.23	0.00
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.10	1.6	0.2903	1	19.23	0.00
	C Powell pers comm 2001	Chepiwanoxet average 1990-2000 (seine)	0.09	2.0	0.2903	1	1.00	0.05
	C Powell pers comm 2001	Wickford average 1990-2000 (seine)	0.20	2.0	0.2903	1	1.00	0.12
	J. Temple pers comm 2002	Narrow River average 1998-2001 (seine)	0.32	2.0	0.2903	1	1.00	0.19
	J. Temple pers comm 2002	Winnapaug Pond average 1998-2001 (seine)	0.21	2.0	0.2903	1	1.00	0.12
	J. Temple pers comm 2002	Point Judith Pond average 1998-2001 (seine)	0.21	2.0	0.2903	1	1.00	0.12
	Species average							0.09

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Atlantic silverside	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1997	1.23	1.6	0.5022	1	18.18	0.05
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998	0.20	1.6	0.5022	1	18.18	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999	0.07	1.6	0.5022	1	18.18	0.00
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall - July 2000	0.17	1.6	0.5022	1	19.23	0.01
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.07	1.6	0.5022	1	19.23	0.00
	Raposa, in press	Galilee Marsh — 1997	4.78	1.6	0.5022	1	11.90	0.32
	Raposa, in press	Galilee Marsh — 1998	1.73	1.6	0.5022	1	11.90	0.12
	Raposa, in press	Galilee Marsh — 1999	14.38	1.6	0.5022	1	11.90	0.97
	Species average							0.19

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Grubby	no obs.							
Striped killifish	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1997	0.70	1.6	0.5474	1	18.18	0.03
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998	0.17	1.6	0.5474	1	18.18	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999	0.55	1.6	0.5474	1	18.18	0.03
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1998 (lift net)	0.01	1.3	0.5474	1	1.00	0.01
	Roman et al., submitted to <i>Restoration Ecology</i>	Sachuest Point — 1999 (lift net)	0.01	1.3	0.5474	1	1.00	0.01
	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — July 2000	2.40	1.6	0.5474	1	19.23	0.11

Table 4-24. Final estimates of the annual increase in production of age-1 equivalent fish per square meter of restored tidal wetland for Pilgrim species that would benefit most from tidal wetland restoration (cont.).

Species	Source of initial species density estimate	Sampling location and date ^a	Reported/ calculated species density estimate per m ² of tidal wetland	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored habitat service flow adjustment factor	Sampling time and location adjustment factor	Increased production of age 1 fish per m ² of restored tidal wetland ^b
Striped killifish	Raposa pers comm 2001	NERR — Prudence Isl. Coggeshall — Sept. 2000	0.53	1.6	0.5474	1	19.23	0.02
	Raposa, in press	Galilee Marsh — 1997	4.35	1.6	0.5474	1	11.90	0.32
	Raposa, in press	Galilee Marsh — 1998	3.50	1.6	0.5474	1	11.90	0.26
	Raposa, in press	Galilee Marsh — 1999	12.40	1.6	0.5474	1	11.90	0.91
	Species average							0.17
Striped bass	no obs.							
Bluefish	no obs							

a. Sampling results are based on collections using 1 m² throw traps unless otherwise noted.

b. Calculated by multiplying the initial species density estimate by the sampling efficiency, life stage, and restored habitat service flow adjustment factors and dividing by the sampling time and location adjustment factor.

4.5.3.2 Adjusting artificial reef sampling results to estimate annual average increase in production of age-1 fish

As with the other restoration alternatives, sampling efficiency, life stage conversion, and restored versus undisturbed habitat adjustments were made to production estimates for artificial reef habitats. These adjustments are discussed below.

Sampling efficiency

The same sampling efficiency adjustment factor of 2.0 is incorporated for the tautog abundance estimates developed from the Rhode Island juvenile finfish survey as was used in the sampling efficiency adjustments from this survey for winter flounder. The 2.0 adjustment factor represents the bottom range (conservative assumption) of a seine net's sampling efficiency (50%), based on the judgment of the current staff of Rhode Island's coastal pond fish survey (personal communication, J. Temple, Rhode Island Division of Fish and Wildlife, 2002).

The sampling efficiency of the baited traps and tagging procedure used in Lawton et al. (2000) was assumed to be 1.0, as the results of the study already incorporate sampling efficiency as reported.

Conversion to the age-1 equivalent life stage

The information used to develop life stage adjustment factors for juvenile fish to age-1 equivalents is presented in Table 4-28 for the Pilgrim species that would benefit most from artificial reef development.

Table 4-28. Life stage adjustment factors for Pilgrim species — artificial reef.

Species	Oldest life stage before age 1 in I&E model	Estimated survival rate to age 1	Sampled life stage	Estimated life stage adjustment factor
Rock gunnel	larvae	0.1416	juvenile	0.5708
Radiated shanny	larvae	0.0853	juvenile	0.5426
Sculpin spp.	larvae	0.0180	juvenile	0.5090
Tautog	larvae	0.0001	juvenile	0.5001

The Rhode Island juvenile finfish survey primarily captures juvenile tautog. However, the size distribution of cunner suggests that primarily adult fish were captured. Some of these cunner were likely older than age 1. To convert the raw cunner numbers to age-1 equivalents, we used the same factor of 1.39 that is also used in the EAM to convert the raw numbers of cunner impinged to age-1 equivalents.

Adjusting for differences between restored and undisturbed habitats

No available information suggested that artificial reefs are utilized substantially less than natural reefs by the species listed in Table 4-25. Thus, an adjustment factor of 1.0 was incorporated.

4.5.3.3 Final estimates of increases in age-1 production for artificial reefs

Table 4-29 presents the final estimates of annual increased production of age-1 fish, based on the average across all sampling efforts, that would result from artificial reef development for species at Pilgrim.

Table 4-29. Final estimates of annual increased production of age-1 equivalent fish per square meter of artificial reef developed for Pilgrim species.

Species	Source of initial species density estimate	Species abundance estimates (fish/m ² reef)	Sampling efficiency adjustment factor	Life stage adjustment factor	Restored vs. undisturbed habitat adjustment factor	Expected age-1 increased production (fish per m ² artificial reef)
Rock gunnel	no obs.					
Radiated shanny	no obs					
Cunner	Lawton et al. (2000), Plymouth MA	4.06 ^a	1.0	1.39	1.0	5.64
Sculpin spp.	No obs.					
Tautog	RI juvenile finfish survey, 1990-2000: Patience Island	0.028	2.0	0.5001	1.0	0.03
	RI juvenile finfish survey, 1990-2000: Spar Island	0.031	2.0	0.5001	1.0	0.03
	Species average					0.03

a. Average of the central population estimates for the inner and outer breakwaters.

4.5.4 Estimates of Increased Species Production from Installed Fish Passageways

A habitat-based option for increasing the production of anadromous species is to increase their access to suitable spawning and nursery habitat by installing fish passageways at currently impassible barriers (e.g., dams). The anadromous species at Pilgrim that would benefit most from fish passageways are presented in Table 4-30, along with information on their annual average I&E losses for the period 1974-1999.

Table 4-30. Anadromous species at Pilgrim that would benefit most from fish passageways.

Species	Annual average I&E loss of age-1 equivalents	Percentage of annual average I&E loss across all fish species
Rainbow smelt	1,330,022	9.23%
Atlantic herring	29,079	0.20%
Alewife	4,343	0.03%
Blueback herring	703	0.00%
White perch	73	0.00%
Total	1,364,220	9.46%

4.5.4.1 Abundance estimates for anadromous species

No studies provided direct estimates of increased production of anadromous fish attributable to the installation of a fish passageway. Thus, increased production estimates were based on abundance estimates from anadromous species monitoring programs in Massachusetts and Rhode Island, combined with an estimate of the average increase in suitable spawning habitat that would be provided upstream of the current impassible obstacles following the installation of fish passageways.

Anadromous species abundance in Massachusetts and Rhode Island spawning/nursery habitats

Information on the abundance of anadromous species in spawning/nursery habitat in Massachusetts was available only for a select number of alewife spawning runs in the area around the Cape Cod canal, including locations in Massachusetts Bay and Buzzards Bay (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). Alewife abundance information was also available for the spawning runs at the Gilbert Stuart and Nonquit locations in Rhode Island. These runs are almost exclusively alewives, despite being reported as runs of river herring (i.e., blueback herring and alewives; personal communication, P. Edwards, Rhode Island Department of Environmental Management, 2001). The size of these alewife runs and the associated abundance estimates (number of fish per acre) in available spawning/nursery habitat are presented in Table 4-31.

The Mattapoissett system has low spawning habitat utilization by alewives because of continuing recovery of the system (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). Therefore, the Mattapoissett River values were omitted. This raised the production estimates for fish passageways and reduced the restoration costs for implementing sufficient fish passageways.

Table 4-31. Average run size and density of alewives in spawning nursery habitats in select Massachusetts waterbodies.

Waterbody	Average alewife run size (number of fish)	Average number of fish per acre of spawning/nursery habitat
Back River (MA) (12 year average)	373,608	766
Mattapoissett River ^a (12 year average)	66,457	90
Monument River (MA) (12 year average)	367,521	811
Nonquit system (RI) (1999-2001 average)	192,173	951
Gilbert Stuart system (RI) (1999-2001 average)	311,839	4,586
Average across all sites presented		1,441
Average without Mattapoissett River		1,778

a. The Mattapoissett River is currently in recovery and production has been increasing in recent years (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001).

Average size of spawning/nursery habitat that would be accessed with the installation of fish passageways

Anadromous fisheries staff in Massachusetts revealed that approximately 5 acres of additional spawning/nursery habitat would become accessible for each average passageway installed (personal communication, K. Reback, Massachusetts Division of Marine Fisheries, 2001). This estimate reflects that previous projects have already provided access to most of the available large spawning/nursery habitats.

4.5.4.2 Adjusting anadromous run sampling results to estimate annual average increase in production of age-1 fish

As with the other restoration alternatives, a number of adjustment factors were considered. However, information was much more limited upon which to base these adjustments. Adjustments to convert returning alewives to age-1 equivalents and to account for sampling efficiency were assumed to be 1.0 because of a lack of information. In addition, nothing suggested a basis for adjustments based on differences between existing and new spawning habitat accessed via fish passageways. As a result, an adjustment factor of 1.0 was used.

4.5.4.3 Final estimates of annual age-1 equivalent increased species production

The density of anadromous species in their spawning/nursery habitat, the average increase in spawning/nursery habitat from installation of fish passageways, and adjustment factors are presented in Table 4-32.

Table 4-32. Estimates of increased age-1 fish for Pilgrim species that would benefit most from installation of fish passageways.

Species	Source of initial species density estimate	Species density estimate in spawning/nursery habitat (fish per acre)	Number of additional spawning/nursery habitat acres per new passageway	Life stage adjustment factor	New vs. existing habitat adjustment factor	Calculated annual increase in age-1 fish per new passageway installed ^a
Rainbow smelt	no obs					
Atlantic herring	no obs					
Alewife	Mattapoissett River — (K. Reback MA DMF pers. comm, 2001)	90	5	1	1	452
	Monument River — (K. Reback MA DMF pers. comm, 2001)	810	5	1	1	4,054
	Back River — (K. Reback MA DMF pers. comm, 2001)	766	5	1	1	3,828
	Nonquit river system — (P. Edwards, RI DEM, pers comm, 2001)	951	5	1	1	4,757
	Gilbert Stuart river system — (P. Edwards, RI DEM, pers comm, 2001)	4,586	5	1	1	22,929
	Species average (excluding Mattapoissett River)^b					8,892
Blueback herring	no obs.					
White perch	no obs.					

a. This value is the product of the values in the five data fields.

b. As previously noted, the Mattapoissett results are excluded in calculating the species average for alewife because the low density estimates are attributable to the system recovering from previous stressors.

4.5.5 Estimates of Increase in Age-1 Fish Production from Water Quality Improvements or Reduced Fishing Pressure

Resource managers and restoration experts indicated that a number of Pilgrim species would benefit most from improved water quality or reduced fishing pressure because they met at least one of the following criteria:

- ▶ The species is pelagic (e.g., Atlantic menhaden).
- ▶ There is no obvious habitat that the species prefers or relies on that could be practically restored (e.g., hogchoker).
- ▶ The preferred habitat is in deep water (e.g., greater than 30 feet) or very deep water (e.g., greater than 100 feet), which limits practical options for habitat restoration because of cost or technical constraints (e.g., fourbeard rockling, American plaice).

As a result, pursuing improvements in water quality and/or reducing fishing pressure were selected as the preferred restoration alternatives for these species. The species at Pilgrim that would benefit most from improving water quality or reducing fishing pressure are listed in Table 4-33, along with annual average I&E losses for the period 1974-1999.

Table 4-33. Pilgrim species that would benefit most from improving water quality or reducing fishing pressure.

Species	Average annual I&E loss of age-1 equivalent organisms	Percentage of total I&E losses for all species
Finfish		
Fourbeard rockling	411,191	2.85%
Windowpane	17,542	0.12%
Atlantic menhaden	14,270	0.10%
Atlantic mackerel	6,662	0.05%
Searobin	3,767	0.03%
Red hake	1,774	0.01%
Lumpfish	1,297	0.01%
Butterfish	399	0.00%
American plaice	221	0.00%
Scup	114	0.00%
Little skate	78	0.00%
Bay anchovy	18	0.00%
Hogchoker	2	0.00%
Total	457,335	3.17%
Shellfish		
Blue mussels	159,880,528,203	100%

Despite the magnitude of I&E losses for these species, and the fact that improving water quality and reducing fishing pressure would benefit all species to varying degrees, it was beyond the scope of this HRC to develop quantitative estimates of the increased production of age-1 fish from these two alternatives. This reflects both budget constraints and a lack of readily available information describing how much water quality projects would improve water quality, and how much water quality improvements would increase fish production. In addition, significant uncertainty exists regarding the effectiveness of nonregulatory actions that could be undertaken to reduce fishing pressure. The limits to developing quantitative estimates of the increased production of age-1 fish are reviewed in the following subsections.

4.5.5.1 Limits to quantifying age-1 production increases from water quality improvements

Several actions could improve water quality without transferring legal responsibility from one party to another. For example, buffer strip development along waterways and septic system improvements would reduce loadings of suspended solids and nutrients into water bodies, improving turbidity, dissolved oxygen content, and chemical concentrations. These improvements could be linked to increases in age-1 fish directly by reducing mortality, or indirectly by stimulating increased natural production.

The expected average annual increases in fish production associated with these restoration actions were not quantified because developing or interpreting complex water quality, concentration-response, and population models was beyond the scope of this HRC valuation. However, these relationships could be developed with additional time and effort.

4.5.5.2 Limits to quantifying increased species production from reduced fishing pressure

Most actions that can achieve lasting reductions in fishing pressure require changes in existing regulations. However, regulatory changes were beyond the scope of this HRC valuation, particularly because of the uncertainty concerning the lack of established property rights for individual fish. Absent these rights, which could be established through individual allocations of a fixed quota on commercial and recreational catches, reducing fishing pressure on a species generally involves persuading current participants in the fishery to cease or reduce their operations.

While market-based programs such as commercial boat buy-backs (Kitts and Thunberg, 1998) have been implemented to reduce fishing pressure, their impact is uncertain because these boats generally have an operating license that permits a limited number of days at sea or other level of effort. While this limits the number of days at sea for a given fleet, its impact may be minimal if the most productive boats remain in the fleet. Further, removing the effort of a given boat may have little impact if it was not actively fishing or if the remaining vessels increase their level of effort. For these reasons the potential benefits of reduced fishing pressure were not quantified.

4.6 Step 6: Scaling Preferred Restoration Alternatives

The following subsections calculate the required scale of implementation for each of the preferred restoration alternatives for each species. The quantified I&E losses are divided by the estimates of the increased fish production, giving the total amount of each restoration needed to offset I&E losses for each species.

4.6.1 SAV Scaling

The information used to scale SAV restoration is presented in Table 4-34.

Table 4-34. Scaling of SAV restoration for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per 100 m ² of revegetated substrate (rounded)	Number of 100 m ² units of revegetated SAV required to offset estimated average annual I&E loss
Atlantic tom cod	2,439	0.99	2,475
Pollock	525	no obs.	N/A
Northern pipefish	118	2.50	47
Threespine stickleback	118	19.29	6
Required units of implementation to offset I&E losses across species			2,475

4.6.2 Tidal Wetlands Scaling

The information used to scale tidal wetland restoration is presented in Table 4-35.

Table 4-35. Scaling of tidal wetland restoration for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per m ² of restored tidal wetland (rounded)	Number of m ² units of restored tidal wetland required to offset estimated average annual I&E loss ^a
American sand lance	4,116,285	no obs.	N/A
Winter flounder	210,715	0.09	2,429,812
Atlantic silverside	25,929	0.19	139,539
Grubby	879	no obs.	N/A
Striped killifish	90	0.17	527
Striped bass	9	no obs.	N/A
Bluefish	2	no obs.	N/A
Required units of implementation to offset I&E losses across species			2,429,812

a. A restored wetland area refers to an area in a currently restricted tidal wetland where invasive species (e.g., *Phragmites* spp.) have overtaken salt tolerant tidal marsh vegetation (e.g., *Spartina* spp.) and that is expected to revert to typical tidal marsh vegetation once tidal flows are returned. Waterways adjacent to these vegetated areas are also included in calculating the potential area that could be restored in a tidal wetland.

4.6.3 Reef Scaling

The information used to scale artificial reef development is presented in Table 4-36.

Table 4-36. Scaling of artificial reef development for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per m ² of artificial reef (rounded)	Number of m ² units of artificial reef surface habitat required to offset estimated average annual I&E loss
Rock gunnel	4,862,872	no obs.	N/A
Radiated shanny	1,644,456	no obs.	N/A
Cunner	993,911	5.64	176,218
Sculpin species	734,773	no obs.	N/A
Tautog	1,076	0.03	36,699
Required units of implementation to offset I&E losses across species			176,218

4.6.4 Anadromous Fish Passage Scaling

The information used to scale fish passageway installation is presented in Table 4-37.

Table 4-37. Scaling of anadromous fish passageways for Pilgrim species.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish per passageway installed (rounded)	Number of new fish passageways required to offset estimated average annual I&E loss
Rainbow smelt	1,320,022	no obs.	N/A
Atlantic herring	29,079	no obs.	N/A
Alewife	4,343	8,892	0.49
Blueback herring	703	no obs.	N/A
White perch	73	no obs.	N/A
Required units of implementation to offset I&E losses across species			0.49

4.6.5 Water Quality Improvement/Reduce Fishing Pressure Scaling

It was not possible to scale sufficient water quality improvements and reduced fishing pressure to offset I&E losses. The Pilgrim species that would benefit most from improving water quality and reducing fishing pressure are presented in Table 4-38. Scaling this restoration alternative likely would increase the Pilgrim HRC estimate significantly, as discussed in Section 4.9.

Table 4-38. Pilgrim species that would benefit most from improved water quality/reduced fishing pressure.

Species	Average annual I&E loss of age-1 equivalent fish	Best estimate of increased production of age-1 fish from water quality/reduced fishing pressure improvements	Number of units of water quality improvement required to offset estimated average annual I&E loss
Finfish			
Fourbeard rockling	411,191	no obs.	N/A
Windowpane	17,542	no obs.	N/A
Atlantic menhaden	14,270	no obs.	N/A
Atlantic mackerel	6,662	no obs.	N/A
Searobin	3,767	no obs.	N/A
Red hake	1,774	no obs.	N/A
Lumpfish	1,297	no obs.	N/A
Butterfish	399	no obs.	N/A
American plaice	221	no obs.	N/A
Scup	114	no obs.	N/A
Little skate	78	no obs.	N/A
Bay anchovy	18	no obs.	N/A
Hogchoker	2	no obs.	N/A
Shellfish			
Blue mussel	159,880,528,203	no obs.	N/A

4.7 Unit Costs

The seventh step of the HRC valuation is to develop unit cost estimates for the restoration alternatives. Unit costs account for all the anticipated expenses associated with the actions required to implement and maintain restoration. Unit costs also included the cost of monitoring to determine increased production of age-1 fish. Unit costs were expressed as the current level of funding required to cover all expenses over the anticipated project life.

All major project expenditures were assumed to occur in the first year, leaving only maintenance and monitoring expenses in subsequent years. Most of these projects were assumed to require little or no maintenance. The monitoring programs were assumed to last for 10 years. Therefore, the current funding required for a unit of each restoration alternative was calculated as the sum provided at the project outset that could fund all activities for 10 years, accounting for inflation and interest. The following price inflation and interest earnings assumptions were made:

- ▶ An annual price inflation rate of 3.0% was used, consistent with the observed annual rate in the Consumer Price Index from 1990 through 2000 (U.S. Bureau of Labor Statistics, 2001).
- ▶ Interest earnings were calculated by multiplying remaining balances at the end of each year by the estimated December 2001 Treasury bill rate of 5.16% (U.S. Bureau of Housing and Urban Development, 2001).

4.7.1 Unit Costs of SAV Restoration

Unit cost estimates for SAV restoration were expressed as the present value of costs per 100 m² for direct comparison with increased production estimates. A number of completed and ongoing SAV restoration projects were evaluated, and monitoring costs were included. The following subsections describe how implementation and monitoring costs were derived for SAV restoration.

4.7.1.1 Implementation costs

Save the Bay has a long history of SAV habitat assessment and restoration in the Naragansett and Mount Hope Bays. A Save the Bay SAV restoration project begun in the summer of 2001 involved transplanting eelgrass to revegetate 16 m² of habitat at each of three sites in Naragansett Bay. Cost information from this project was used to develop unit cost estimates for implementing SAV restoration per 100 m² of revegetated habitat.

Save the Bay's cost proposal estimated that \$93,128 (2001 dollars) would be required to collect and transplant eelgrass shoots over 48 m² of revegetated habitat. These costs include collecting and transplanting the SAV shoots to provide an initial density of 400 shoots per revegetated square meter of substrate. Averaged over the 48 m² of habitat being revegetated, this provides an average unit cost of \$1,940 per m². The unit costs comprise the following categories:

- ▶ labor: 70.7% (includes salaried staff with benefits, consultants, and accepted rates for volunteers)
- ▶ boats: 15.2% (expenses for operating the boat for the collecting and transplanting)
- ▶ materials and equipment: 9.6%
- ▶ overhead: 4.6% (calculated as a flat percentage of the labor expenses for the salaried staff).

Contingency expenses were set at 10% (\$194 per m²). The costs of identifying and evaluating the suitability of potential restoration sites were set at 1% (\$19 per m²). No costs were added for maintaining the service flows provided by the project, because SAV restoration requires little direct maintenance. This reflects both the relative inaccessibility of SAV sites and the relative importance of factors beyond direct control, such as local water quality and extreme weather.

Costs were also adjusted to account for natural growth and spreading from the original transplant sites to the bare spots between transplants (Short et al., 1997). For example, Dr. Frederick Short (University of New Hampshire's Jackson Estuarine Laboratory) planted between 120 and 130 TERFS (Transplanting Eelgrass Remotely with Frame Systems), each 1 m², in each acre of seabed to be revegetated at a SAV restoration site (personal communication, P. Colarusso, U.S. EPA Region 1, 2002). Assuming complete coverage over time, this results in a ratio of plantings to total coverage of between 1:31 (130 1 m² TERFS / 4,047 m² per acre) and 1:34 (120 1 m² TERFS / 4,047 m² per acre).

However, the initially bare areas do not revegetate immediately. Therefore, an assumption was made that the area covered would double each year. Under this assumption, the entire area would be filled in the sixth year of the restoration project. Using the habitat equivalency analysis (HEA) method (Peacock, 1999), the present value of the services over the 6 years is 90% of that provided by instantaneous revegetation. Therefore, 90% of the 1:34 planting-to-coverage ratio, or 1:30, was applied. Table 4-39 presents the components of implementation unit cost for SAV restoration, incorporating the adjustment ratio in the last step.

Table 4-39. Implementation unit costs for SAV restoration.

Expense category	Cost per m ² of SAV restored	Cost per 100 m ² of SAV restored
Direct restoration (shoot collection and transplant)	\$1,940	\$194,000
Contingency costs (10% of direct restoration)	\$194	\$19,400
Restoration site assessment (1% of direct restoration)	\$19	\$1,900
Subtotal without allowance for distribution of transplanted SAV shoots	\$2,154	\$215,400
Discounted rate of return on transplanted SAV	30:1	30:1
Final implementation unit costs	\$71.80	\$7,180

4.7.1.2 Monitoring costs

SAV restoration monitoring improves the inputs to the HRC analysis by quantifying the impact of the SAV restoration on fish production/recruitment in the restoration area, and the rate of growth and expansion of the restored SAV bed. The most efficient way to achieve both of these goals would be for divers to evaluate the number of adult fish in the habitat and the vegetation density, combined with throw trap or drop trap sampling of juvenile fish using the habitat (Short et al., 1997). Diver-based monitoring minimizes damage to sites, expands the areas that can be sampled, and increases sampling efficiency compared to trawl-based monitoring (personal communication, J. Hughes, NOAA Marine Biological Laboratory, 2001).

Hourly rates for the divers and captain were provided by Save the Bay (personal communication, A. Lipsky, Save the Bay, 2001), and the daily rate for the boat was based on rate information from NOAA's Marine Biological Laboratory in Woods Hole (personal communication, J. Hughes, NOAA, 2001). Because SAV monitoring costs will be significantly affected by the size, number, and distance between restored SAV habitats, large areas can be covered in a single day only when continuous habitats are surveyed. Smaller, disconnected habitats will require much more time to cover. Therefore, total monitoring costs are somewhat unpredictable and were assumed to be equal to initial revegetation costs. This simplifying assumption is neither conservative, nor liberal. The summary of the available SAV monitoring costs and the final assumption used are presented in Table 4-40.

Table 4-40. Estimated annual unit costs for a SAV restoration monitoring program.

Annual expenditures			
Expense category	Quantity	Daily rate	Total cost
Monitoring crew	3 (2 divers and boat captain/assistant)	\$268	\$804
Monitoring boat	1	\$150	\$150
Total daily rate			\$954
Assumed PV cost for SAV monitoring per 100 m ² restored habitat			\$7,180

4.7.1.3 Total SAV restoration costs

Combining the unit costs for restoration and monitoring, the cost for a 100 m² unit of SAV restoration for 10 years is \$14,360.

4.7.2 Unit Costs of Tidal Wetland Restoration

Many different actions may be needed to restore flows to a wetland site, and project costs can vary widely. These issues are addressed in the following subsections, which present the development of the unit costs for tidal wetland restoration.

4.7.2.1 Implementation costs

Costs for restoration of tidally restricted marshes depend heavily on the type of restriction that is impeding tidal flow into the wetland. Possible sources of the restriction in tidal flow include improperly designed or located roads, railroads, bridges, and dikes, all of which can eliminate tidal flows or restrict tidal flows via improperly sized openings. A compilation of tidally restricted salt marsh restoration projects in the Buzzards Bay watershed (Buzzards Bay Project National Estuary Program, 2001) describes restrictions and costs to return tidal flows to over 130 sites. These cost estimates include expenses for project design, permitting, and construction, and are estimated on a predictive cost equation that was fitted from the actual costs and budgets for a limited number of projects (Buzzards Bay Project National Estuary Program, 2001).

Staff involved in the Buzzards Bay assessment provided the current project database, which includes the following information (personal communication, J. Costa, Buzzards Bay National Estuary Program, 2001):

- ▶ nature of the tidal restriction
- ▶ estimated cost to address the tidal restriction
- ▶ size of the affected tidal wetland (in acres)
- ▶ acreage of the *Phragmites* in the tidally restricted wetland.

Some of the project costs used in the cost estimation equation were provided by public agencies, which were lower than market prices (personal communication, J. Costa, Buzzards Bay National Estuary Program, 2001). Therefore, the cost estimates were adjusted upward by a factor of 2.0, consistent with the adjustment recommended in the report (Buzzards Bay Project National Estuary Program, 2001). The adjusted total project costs were then divided by the acres of *Phragmites* in the wetland to provide the cost per acre (sites with no *Phragmites* were eliminated from consideration). Table 4-41 summarizes costs based on the cost factor (an input in the cost estimation equation), type of restriction found at the site, and the number of *Phragmites* acres at the location. An alternative summary of these projects is presented in Table 4-42, where the projects are organized by acres of *Phragmites* at the site, not the current tidal restriction.

Combined, Tables 4-41 and 4-42 show significant variability in the per acre costs for tidal wetland restoration. Therefore, the median cost of \$71,000 per acre of tidal wetland restoration was used. Table 4-43 presents the final per acre implementation costs for tidal wetland restoration. These costs include the median per acre restoration cost, \$750 per acre, paid by the Rhode Island Department of Environmental Management's Land Acquisition Group for this type of land (personal communication, L. Primiano, Rhode Island Department of Environmental Management, 2001).

Table 4-41. Salt marsh restoration costs.

Restriction structure class	Cost factor	Phragmites acres	Number of sites	Cumulative Phragmites acreage	Average Phragmites acreage	Total private cost	Average cost per Phragmites acre restored (from total cost and acres)	Minimum cost per Phragmites acre restored	Maximum cost per Phragmites acre restored
culvert	0.5	acres < 1	16	6.59	0.41	\$335,357	\$50,889	\$17,921	\$578,081
culvert	0.5	1 < acres < 5	11	20.37	1.85	\$242,496	\$11,903	\$3,242	\$71,045
culvert	0.5	5 < acres < 10	1	8.56	8.56	\$20,825	\$2,434	\$2,434	\$2,434
dike	0.5	acres < 1	1	0.35	0.35	\$13,211	\$38,073	\$38,073	\$38,073
road	0.5	1 < acres < 5	1	1.67	1.67	\$19,116	\$11,447	\$11,447	\$11,447
culvert	1	acres < 1	31	13.26	0.43	\$1,797,450	\$135,585	\$21,518	\$10,490,647
culvert	1	1 < acres < 5	23	46.02	2.00	\$1,225,745	\$26,633	\$5,312	\$84,770
culvert	1	5 < acres < 10	2	16.43	8.22	\$248,878	\$15,144	\$9,898	\$22,608
culvert	1	10 < acres < 25	2	41.97	20.99	\$91,451	\$2,179	\$1,919	\$2,449
dike	1	10 < acres < 25	1	12.00	12.00	\$6,053,000	\$504,417	\$504,417	\$504,417
fill	1	acres < 1	1	0.12	0.12	\$31,142	\$251,146	\$251,146	\$251,146
road	1	acres < 1	1	0.10	0.10	\$29,396	\$293,958	\$293,958	\$293,958
road	1	1 < acres < 5	1	2.31	2.31	\$35,231	\$15,265	\$15,265	\$15,265
wall	1	acres < 1	2	0.96	0.48	\$148,819	\$154,697	\$25,661	\$5,936,752
bridge	3	acres < 1	8	5.12	0.64	\$21,208,029	\$4,140,576	\$184,170	\$13,418,293
bridge	3	1 < acres < 5	12	27.32	2.28	\$27,704,691	\$1,014,192	\$184,048	\$3,663,062
bridge	3	5 < acres < 10	2	11.01	5.51	\$6,606,000	\$599,946	\$399,746	\$800,545
bridge	3	10 < acres < 25	8	103.49	12.94	\$92,094,000	\$889,883	\$56,300	\$3,300,250
bridge	3	25 < acres < 50	4	157.28	39.32	\$8,262,000	\$52,529	\$22,882	\$105,968
bridge	3	50 < acres	1	113.00	113.00	\$6,163,000	\$54,540	\$54,540	\$54,540
railroad	4	acres < 1	1	0.41	0.41	\$66,841	\$163,826	\$163,826	\$163,826
railroad	4	1 < acres < 5	3	3.61	1.20	\$1,078,692	\$298,476	\$208,033	\$13,418,293

Table 4-42. Average per acre cost of restoring *Phragmites* in Buzzards Bay restricted tidal wetlands.

<i>Phragmites</i> acres	Number of sites	Cumulative acreage	Average acreage	Total private cost	Average cost per <i>Phragmites</i> acre restored (from total cost and acres)
acres < 1	61	26.91	0.44	\$23,630,245	\$878,121
1 < acres < 5	51	101.31	1.99	\$30,305,971	\$299,153
5 < acres < 10	5	36.00	7.20	\$6,875,703	\$190,992
10 < acres < 25	11	157.46	14.31	\$98,238,451	\$623,895
25 < acres < 50	4	157.28	39.32	\$8,262,000	\$52,529
50 < acres	1	113.00	113.00	\$6,163,000	\$54,540
Total	133	591.96	4.45	\$173,475,370	\$293,053
Median					\$71,000

Table 4-43. Implementation unit costs for tidal wetland restoration incorporated in the HRC.

Implementation cost description	Source of estimate	Value (2001 dollars)
Restore tidal flows to restricted areas	Median of adjusted costs from Buzzards Bay project database	\$71,000
Acquire tidal wetlands	Midpoint of range of paid for tidal wetlands by Rhode Island DEM	\$750

4.7.2.2 Monitoring costs

Neckles and Dionne (1999) present a sampling protocol, developed by a workgroup of experts, for evaluating nekton use in restored tidal wetlands. The sampling plan calls for different sampling techniques and frequencies to capture fish of various sizes in both creek and flooded marsh habitats of a tidal wetland. A summary of these recommendations is presented in Table 4-44.

Table 4-44. Sampling guidelines for nekton in restored tidal wetlands.

Sampling location	Sampling technique	Sampling time	Sampling frequency
Creeks (for small fish)	Throw traps	midtide during spring tide cycle	2 dates in August
Creeks (for larger fish)	Fyke net	slack tide during spring tide cycle	2 dates in August (same as for throw trap work) and 2 dates in spring
Flooded wetland surface	Fyke net	spring tide cycle	1 date in August

Source: Neckles and Dionne, 1999.

The sampling protocol suggests that one technician and two volunteers can provide the necessary labor. The estimated annual cost in the first year of monitoring is \$1,600. This cost comprises \$490 in labor for the three workers over 5 days (3 in August and 2 in the spring, with 8-hour days, \$15 per hour for volunteers, and \$30 per hour for the technician). The \$1,100 in equipment costs includes two fyke nets and two throw traps at \$500 for the fyke nets and \$50 for homemade throw traps (Neckles and Dionne, 1999). Two sets of this sampling equipment would allow simultaneous sampling in a restored marsh and at a reference location. Treating these costs as a per acre cost for aggregation with implementation costs probably overstates the frequency of sampling required at the site. However, the initial year labor cost of \$500 per acre has little impact compared to implementation and overall costs.

4.7.2.3 Total tidal wetland restoration costs

Combining implementation and monitoring costs for tidal wetland restoration with annual price inflation (3%) and interest earned on balances carried over (5.16%), the cost for an acre of tidal wetland restoration is \$78,500, or \$19 per m², which was used in the development of the total Pilgrim HRC valuation.

4.7.3 Artificial Reef Unit Costs

The unit cost estimates for developing and monitoring artificial reefs are based the construction and monitoring of six 30 ft x 60 ft reefs constructed of 5-30 cm diameter stone in Dutch Harbor, Naragansett Bay (personal communication, J. Catena, NOAA Restoration Center, 2001). While these reefs were constructed for lobsters, surveys of the Dutch Harbor reef have noted abundant fish use of the structures (personal communication, K. Castro, University of Rhode Island, 2001).

4.7.3.1 Implementation costs

The summary cost information for the design and construction of the six reefs in Dutch Harbor is presented in Table 4-45 (personal communication, J. Catena, NOAA Restoration Center, 2001).

Table 4-45. Summary cost information for six artificial reefs in Dutch Harbor, Rhode Island.

Project component	Cost
Project design	not explicitly valued, received as in-kind services
Permitting	not explicitly valued, received as in-kind services
Interagency coordination	not explicitly valued, received as in-kind services
RFP preparation	not explicitly valued, received as in-kind services
Contract management	not explicitly valued, received as in-kind services
Baseline site evaluation	\$12,280
Reef materials (600 yd ³ of 2-12 in. stone)	\$12,000
Reef construction	\$35,400
Total	\$59,680

These costs were converted to cost per square meter of surface habitat. The cumulative surface area of the six reefs, assuming that the reefs have a sloped surface on both sides, and based on the volume of material used, is approximately 1,024 m². Dividing the total project costs by this surface area results in an implementation cost of \$58/m² of artificial reef habitat.

4.7.3.2 Monitoring costs

Monitoring costs for the Dutch Harbor reefs were \$140,000 over a 5 year period. Again, assuming similar assessment techniques would be required to evaluate fish use and production of an artificial reef (i.e., diver surveys and trap work), these costs are adjusted to provide a monitoring expense of \$28,000.

4.7.3.3 Total artificial reef costs

Combining costs for implementation and monitoring of an artificial reef with annual price inflation (3%) and the interest earned on balances carried over (5.16%), the cost is \$308/m² (\$315,167/1,024 m² surface area over the six reefs), which was used in the development of the total Pilgrim HRC valuation.

4.7.4 Costs of Anadromous Fish Passageway Improvements

Unit costs for fish passageways were developed from a series of budgets for prospective anadromous fish passageway installation, combined with information provided by staff involved with anadromous species programs in Massachusetts and Rhode Island. The implementation, maintenance, and monitoring costs for a fish passageway are presented in the following subsections.

4.7.4.1 Implementation costs

Projected costs for four new Denil type fish passageways on the Blackstone River at locations in Pawtucket and Central Falls, Rhode Island, provide the base for the implementation cost estimates for anadromous fish passageways (personal communication, T. Ardito, Rhode Island Department of Environmental Management, 2001). The reported lengths of the passageways in these projects ranged from 32 m to 82 m, with associated changes in vertical elevation ranging from slightly more than 4 m to approximately 10 m based on the reported slope ratios of 1:8.

The average cost for these projects was \$513,750. The average cost per meter of passageway length was \$10,300 and per meter of vertical elevation covered was \$82,600. These estimates are consistent with the approximate values of \$9,800 per meter of passageway length and \$98,000 per vertical meter suggested by the U.S. Fish and Wildlife Service's regional Engineering Field Office (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). An alternative style of fish passageway, the Alaskan steep, has lower unit costs of \$33,000 per vertical meter, but is not suited for many locations. Therefore, its costs were not used to develop implementation unit cost estimates. While all parties contacted noted that fish passageway costs are extremely sensitive to local conditions, this HRC valuation uses the estimate of \$513,750 as its basic implementation unit cost for installing an anadromous fish passage, assuming the characteristics of the four sites on the Blackstone River are representative of the conditions that would be found at other suitable locations for new passageways.

4.7.4.2 Maintenance and monitoring costs

Maintenance requirements for the Denil fish passageway are minimal and generally consist of periodic site visits to remove any obstructions, typically with a rake or pole (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). Denil passageways located in Maine are still functioning after 40 years, so no replacement costs were considered as part of the maintenance for the structure. Monitoring a fish passageway consists of installing a fish counting monitor and retrieving its data.

A new fish passageway would be visited three times a week during periods of migration (personal communication, D. Quinn, U.S. Fish and Wildlife Service, 2001). Each site visit would require 2 hours of cumulative time during 8 weeks of migration. Volunteer labor costs \$15/hr. Therefore, the annual cost for labor in the first year would be \$740. The cost of a fish counter is \$5,512, based on the average price of two fish counters listed by the Smith-Root Company (Smith-Root, 2001).

4.7.4.3 Total fish passageway unit costs

Combining the costs for implementation, maintenance, and monitoring of an anadromous fish passageway with the annual price inflation (3%) and the interest earned on balances carried over (5.16%), the cost of a single new Denil type fish passageway is \$526,000.

4.7.5 Unit Costs for Water Quality Improvements/Reductions in Fishing Pressure

Because increased fish production from water quality improvements or reduced fishing pressure was not calculated, unit costs were not determined for this restoration option. However, examples of water quality improvement projects were summarized to provide a sense of the potential magnitude of costs. The costs of a commercial boat buyback program to reduce fishing pressure on various Northeast groundfish stocks were also summarized. The cost summaries are presented in the following subsections.

4.7.5.1 Cost information from a select set of water quality improvement projects

Table 4-46 provides information from several water quality improvement projects in coastal areas between Massachusetts Bay and Narragansett Bay that address nutrient and bacterial pollution resulting from sanitary waste and other anthropogenic sources. Table 4-46 also shows a wide range of water quality projects involving a wide range of water quality impacts. These projects represent only a few of the projects that could improve water quality in the waters from Massachusetts Bay to Narragansett Bay. Existing project proposals could easily cost billions of dollars.

Table 4-46. Examples of nonpoint source pollution restoration projects in Massachusetts.

Project	Location	Goals	Tasks	Total Cost
Combined sewer overflow (CSO) upgrade ^a	Naragansett Bay, Providence, Pawtucket, and Central Falls, RI	Treatment of ~2.2 billion gallons of waste that are discharged untreated into the bay each year from the combined sewer overflows.	Construct 6 miles of underground storage tunnels, two sedimentation/disinfection treatment facilities, one wetland treatment system, and sewer separation of 12 areas.	\$389,000,000
Septic system improvements ^b	Bluefish River, Duxbury, MA	Opened soft-shelled clam beds over approximately one-half mile of the river to shellfishing.	Connected septic systems from 3 historic homes and 19 commercial properties on the river to a centralized leach field outside of the river basin.	\$800,000
Stormwater treatment ^c	Onset Bay, Wareham, MA	Part of a series of water quality improvement projects aimed at upgrading seasonally closed shellfishing areas and reducing discharges along public beaches.	Design and construct stormwater remediation best management practices (BMPs) for four stormwater outfalls. Develop a quality assurance plan and perform pre- and post-construction water quality monitoring. Conduct public outreach programs and workshops.	\$218,000
Treatment of road runoff ^c	Three Bay Area/Ropes Beach, Barstable, MA	Protection of Cotuit Bay, a shellfishing area, and gateway to two anadromous fish runs, from nutrient and sediment loading.	Design and install sediment removal tanks, an infiltration system, and a series of rock filled pools and channels to remove sediment bacteria and nitrogen from road runoff contributing to contamination of Cotuit Bay. Develop a quality assurance plan and conduct monitoring. Conduct a technology transfer presentation.	\$157,050
Stormwater treatment ^c	First Herring Brook, Scituate, MA	Protect a pond that supplies the town's water supply from contamination.	Disconnect 9 stormwater discharges in a highly developed area and install infiltration BMPs. Develop a quality assurance plan and conduct monitoring. Make system design to other local developers.	\$129,300
Parking lot runoff treatment ^c	Shaw's Plaza, Sharon, MA	Improve water quality in Billing's Brook and in nearby wetlands and public water supply wells.	Develop and implement stormwater BMPs, including a drainage system with an oil/gas separator catch basin and infiltrations. Develop a maintenance program to ensure that it functions properly. Initiate a public education program on the potential impacts of pollution from runoff from roads and parking lots.	\$48,000

a. NBC, 2001.

b. Personal communication, Joe Grady, Town of Duxbury, 12/07/01.

c. MADEP, 2000.

4.7.5.2 Cost information for commercial boat buyback program

A demonstration of a commercial boat buyback program was conducted in the Northeast groundfish fishery. Permit-holding boat owners were asked to submit a price at which they would be willing to retire their vessel from fishing and relinquish all their existing fishing permits (Kitts and Thunberg, 1998). These bids were then ranked in ascending order based on the ratio of their bid to the groundfish revenue from reported landings by the boat to maximize the impact of the program (i.e., remove the productive boats first).

From June 1995 through May 1998, 79 boats were bought out and retired from commercial fishing at an average price of roughly \$309,000, with a range from \$50,000 to \$1.1 million (Kitts and Thunberg, 1998). On average, permits that allocated 152.9 days at sea per boat, although the average boat was only using 111.8 of these days (Kitts and Thunberg, 1998). The impact of this program on increased production was not quantified.

4.8 Total Cost Estimation

The eighth and final step in the HRC valuation is to estimate the total cost for the preferred restoration alternatives by multiplying the required scale of implementation for each restoration alternative by the complete unit cost for that alternative. The cost of each restoration alternative was sufficient to offset the I&E losses of all Pilgrim species that benefit most from that alternative (i.e., each restoration type was sufficient to offset the single species with the greatest restoration need for that preferred restoration; however, the restoration needs of all species preferring that habitat were not summed because the same habitat benefits each of the species simultaneously). The costs of each restoration program were then summed to determine the total HRC necessary to offset all Pilgrim losses (i.e., multiple restoration programs were required to benefit the diverse species lost at Pilgrim).

The total HRC estimates for the Pilgrim facility are provided in Table 4-47, along with the species requiring the greatest level of implementation of each restoration alternative to offset I&E losses. The scale of implementation, unit costs, and total costs in this table have been rounded to two significant digits to avoid false precision. Resulting total costs also carry two significant digits. These costs can be converted to annualized values by specifying a time period and interest rate.

Table 4-47. Total HRC estimates for Pilgrim I&E losses.

Preferred restoration alternative	Species requiring the greatest level of restoration implementation		Required units of restoration implementation	Units of measure for preferred restoration alternative	Unit cost	Total cost
	Species	Average annual I&E loss of age-1 equivalents				
Improve water quality/reduce fishing pressure	Fourbeard rockling	411,191	N/A	N/A	N/A	N/A
	Blue mussel	159 billion	N/A	N/A	N/A	N/A
Install fish passageways	Alewife	4,343	0.49	new fish passageway	\$530,000	\$530,000 ^a
Create artificial reefs	Cunner	993,911	180,000	m ² of reef surface area	\$310	\$56,000,000
Restore SAV	Atlantic cod	2,439	2,500	100 m ² of directly revegetated substrate	\$14,000	\$35,000,000
Restore tidal wetland	Winter flounder	210,715	2,400,000	m ² of restored tidal wetland	\$19	\$46,000,000
Total HRC						\$140,000,000

a. Anadromous fish passageways must be implemented in whole units, and increased production data are lacking for most affected anadromous species. Therefore, one new passageway was assumed to be warranted.

4.9 Conclusions

HRC analyses indicate that the present value of minimizing I&E at the Pilgrim CWIS is at least \$140 million. This value is significantly greater than the \$6-7 million (7% interest rate, in perpetuity) of foregone recreational and commercial fishing calculated in the Pilgrim case study for EPA's Section 316(b) rule. Recreational and commercial fishing values are lower primarily because they include only a small subset of species, life stages, and human use services that can be linked to fishing. In contrast, the HRC valuation is capable of valuing all species and life stages, and inherently addresses all of the ecological and public services derived from organisms included in the analyses, even when the services are difficult to measure or poorly understood. However, data gaps, time constraints, and budgetary constraints prevented this HRC valuation from addressing most of the aquatic organisms lost to I&E at the Pilgrim facility. In particular, annual losses of 160 billion blue mussels and 460,000 fish comprising 13 species were not included in this HRC valuation, even though water quality improvements are feasible, cost-effective, and most likely able to offset some or all of the I&E losses of these species at Pilgrim. In addition, data gaps for species that were included in the HRC valuation forced many conservative assumptions that most likely underestimated the cost of fully offsetting many I&E losses.

In addition to broadening the species, life stages, and services valued, the Pilgrim HRC valuation provides a roadmap for mitigating I&E losses residual to permitted technologies, and for improving the HRC analyses by closing critical data gaps through effective monitoring. Many of the species experiencing I&E losses at Pilgrim can benefit from tidal wetland, SAV, reef, and fish passage restorations. Careful monitoring of increased production of target species at restoration sites would improve the Pilgrim HRC valuation, and would make HRC valuations at other sites more reliable. Further, HRC restoration monitoring needs align public, Agency, and facility motives. Effective restorations with reliable data can broaden the Agency's analyses of public losses. Effective restorations with reliable data can increase the production of fish per restoration dollar spent by a facility. The public benefits both from additional BTA options justified by more comprehensive valuation and from effective restorations in the natural environment.

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